
LAKESHORE CAPACITY STUDY

TROPHIC STATUS

MAY 1986

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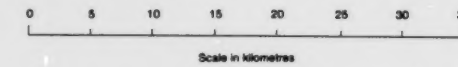
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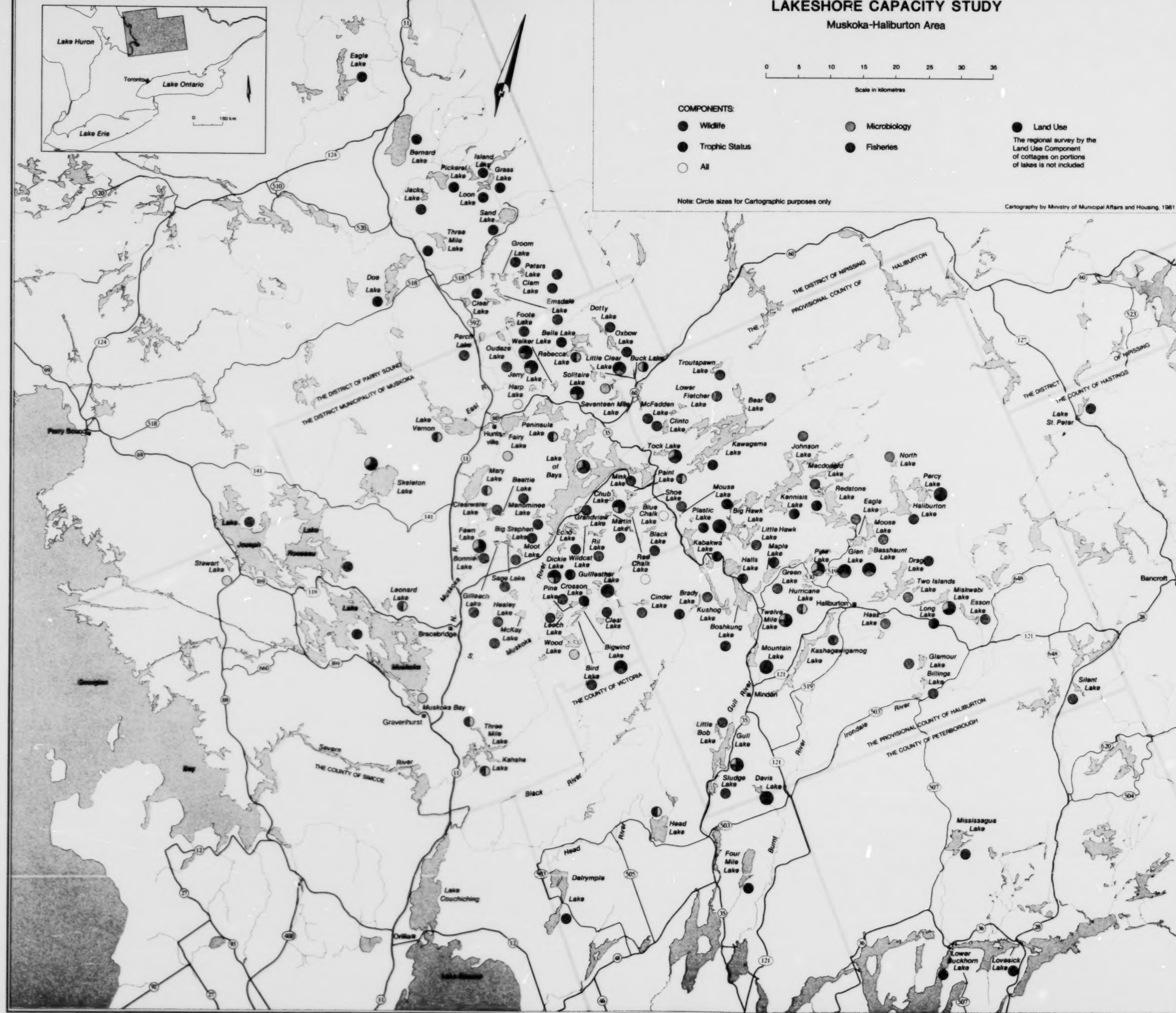
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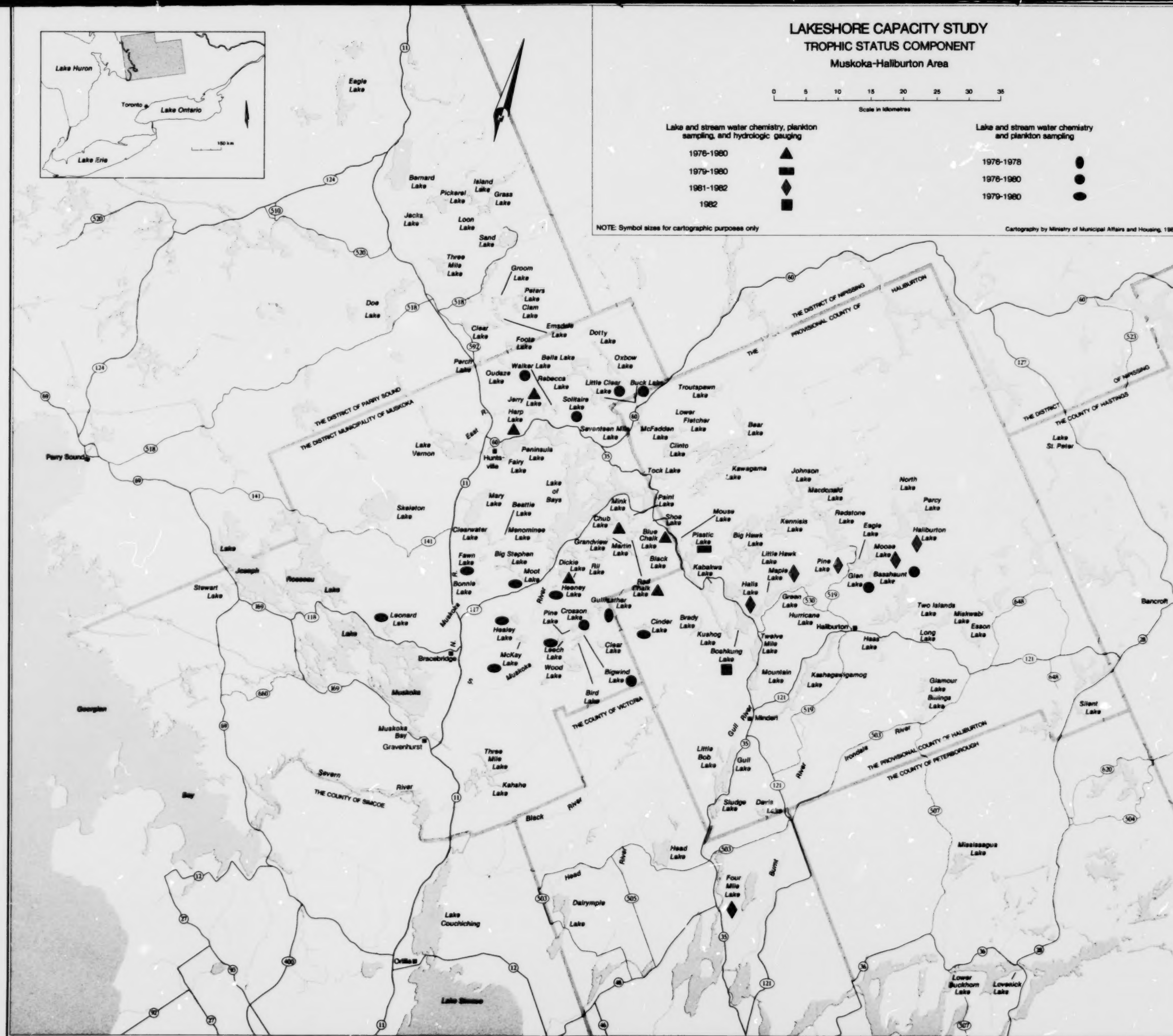
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FOREWORD

Community planners and other professionals involved in the preparation of planning policies for lakeshore development and in the review of specific subdivision proposals have always found it difficult to determine objectively the impact of development on the natural environment. In response to this challenge, the Lakeshore Capacity Study was undertaken to provide a planning tool to assist in evaluating the effects of cottage development on inland lakes and lakeshores. Central to the task was the need to gain a clearer understanding of the relationship between cottage development and its impacts on selected aspects of the natural environment.

To accomplish these objectives, the Ministry of Municipal Affairs carried out the Lakeshore Capacity Study in cooperation with the Ministry of the Environment and the Ministry of Natural Resources.

The Muskoka-Haliburton area of central Ontario was chosen as the Study Area. The homogeneity of the area, which is part of one physiographic region, reduced the need to account for major natural variations among the lakes and watersheds. In addition, the extent of existing development on the lakes varied, permitting an examination of situations ranging from no development to full development.

The Study involved measuring the source of environmental impact, in terms of the lakeshore cottages and their use, and the impact of cottage development in terms of water quality, fisheries, and wildlife habitat. The methods of prediction derived from the research were linked in a simulation model, which is capable of predicting trends in the values of key indicators of impact.

As the research, analysis and findings of the Study are documented in a set of seven reports, selective reading may be desirable. Those readers who have a general interest in the work are advised to read the Committee Report first, as it provides an overall summary of the findings. This should be followed by the Integration report, in which the simulation model is described. Readers with more specialized interests will find the details of each component of the Study in the other five reports. In each report, the reader can select from the table of contents the most important chapters for his or her purposes.

The end product of the Study, the Ontario Lakeshore Capacity Simulation Model, has several features worth noting. The spatial unit addressed by the model is a single lake and the lakeshore. When the model is applied, the number of unknowns related to the natural environment can be substantially reduced, making it easier for planners or other professionals to weigh the environmental effects of development. The model goes a step further to permit

predictions of the impact of cottage development when different management policies are selected.

The scope of the simulation model demands some explanation. In its present form, the model applies to cottage development on inland lakes in the Study Area, where the research was conducted. However, the methods of prediction can be adapted to other parts of the province, as long as differences in conditions are taken into account.

The purpose of the Study was to measure the environmental impact of cottages. Commercial and industrial uses were excluded deliberately, in order to simplify the difficult task of measuring cottage impact. The flexibility inherent in the simulation model makes it possible to add other types of land use later, if so desired.

Most of the existing cottage development in the Study Area is located in a single tier along the shoreline. For this reason, the simulation model applies to the immediate lakeshore and not to backshore development. Again, the methods of prediction developed for cottages near the lake can be adapted to measure the impact of cottage development in other forms.

The model was designed to measure the physical and chemical impacts of cottages. Accordingly, it does not address other planning concerns, such as social and economic impacts. While these were recognized as essential considerations in decision-making, the specific objective of Phase III of the Lakeshore Capacity Study was to find practical ways of producing better technical data regarding environmental impact.

Now that Phase III of the Study is completed, with the production of the Ontario Lakeshore Capacity Simulation Model (OLCSM), the next step envisaged is to apply the model experimentally within the government. In this setting, model output can be tested in a variety of actual development situations. When this period of experimental use has been concluded and the results assessed, the three participating ministries will be able to determine whether the model should be adapted to other parts of the province and whether it should be made available more widely.

The Ministry of Municipal Affairs considers the OLCSM to be a potential planning tool but recognizes that the technical and organizational implications of its use must be examined. While this is underway, the model will be available for testing as an additional planning tool to supplement the information normally required to evaluate a planning policy or development proposal. However, the model will not be used in the decision-making process, which will still rest on the customary range of planning considerations.

This Trophic Status report describes the improved methods of predicting water quality which were developed by scientists in the Ministry of the Environment, as part of Phase III of the Lakeshore Capacity Study.

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GLOSSARY OF SYMBOLS

Symbol	Definition	Units*			
MORPHOMETRY					
A_o	lake area	L^2	L_N	annual natural total phosphorus load	$ML^{-2}T^{-1}$
A_d	catchment or watershed area	L^2	L_A	maximum possible development TP load	$ML^{-2}T^{-1}$
V	lake volume	L^3	L_T	maximum total phosphorus load	$ML^{-2}T^{-1}$
z	measured epilimnetic depth	L	D	grain size measurement	L
$\bar{z}$	lake mean depth	L	H	predicted annual fish harvest	$ML^{-2}T^{-1}$
z_{max}	lake maximum depth	L	E_C	summer fishing effort by cottagers	angler hours
D_L	development of lake length	—	E_n	summer fishing effort by non-residents	angler hours
D_V	development of lake volume	—	E_s	total summer fishing effort	angler hours
D_d	drainage density, stream length per unit drainage area	L^{-1}	E_w	winter fishing effort	angler hours
D_T	depth of thermocline	L	N	number of dwellings	
F	fetch of lake	L	MEI	morphoedaphic index	ML^{-4}
HYDROLOGY AND METEOROLOGY			$\bar{B}$	lake average dry biomass of emergent Diptera	$ML^{-2}T^{-1}$
P	precipitation	L	R_p	retention coefficient, fraction not lost via outflow	—
R	surface runoff into lake	L	R_s	retention capacity	—
G	net ground water gain by the lake	L	S	sedimentation rate measured by lake mass balance	$ML^{-2}T^{-1}$
O	outflow from the lake	L	T	whole-lake sedimentation rate estimate	$ML^{-2}T^{-1}$
E	net evaporation from the lake	L	SAR	sediment accumulation rate	$ML^{-2}T^{-1}$
LE	evaporative heat loss	$Cal\ L^{-2}T^{-1}$	SAR_s	site-specific sediment accumulation rate	$ML^{-2}T^{-1}$
r	net radiation	$Cal\ L^{-2}T^{-1}$	V	sedimentation velocity	LT^{-1}
T	heat storage of the lake	$Cal\ L^{-2}T^{-1}$	A_s	area of the plane through which sedimentation occurs	L^2
B	sensible heat loss of the lake	$Cal\ L^{-2}T^{-1}$	t	time	T
q_s	areal water load	LT^{-1}	PB	phytoplankton biomass	ML^{-3}
Q	lake outlet discharge, lake outflow volume	L^3T^{-1}	CPB	corrected phytoplankton biomass	ML^{-3}
TVA	Tennessee Valley Authority		B_z	total crustacean biomass	ML^{-3}
D	distance	L	B_h	herbivorous crustacean biomass	ML^{-3}
L	stream length	L	FR	zooplankton filtering rates	$L^{-3}animal^{-1}T^{-1}$
OR	organic deposits	%	$AHOD$	areal hypolimnetic oxygen deficit	$ML^{-2}T^{-1}$
ST	shallow till	%	RT	response time	T
E	total phosphorus export	$ML^{-2}T^{-1}$	σ	sedimentation rate constant	T^{-1}
τ_w	water replenishment time	T	ϕ	sum of the loss rate constants = $Q/V + \sigma$	T^{-1}
MODELLING			SUBSCRIPTS		
J_T	total phosphorus input from all sources	MT^{-1}	SO	spring overturn	
J_O	total phosphorus loss from the lake by outflow	MT^{-1}	SS	summer stratified	
J_A	anthropogenic total phosphorus input	MT^{-1}	FO	fall overturn	
$J_{NAT}; J_N$	total input of TP from natural sources	MT^{-1}	IC	ice covered	
J_{PR}	total phosphorus input to lake from precipitation	MT^{-1}	IF	ice free	
L	areal loading rate of total phosphorus	$ML^{-2}T^{-1}$	AN	annual	

EP, E	epilimnion
EM	epi- and metalimnion
H	hypolimnion
OUT	lake outlet
MEAS	measured
N	natural load
T	total load

STATISTICAL SYMBOLS

n	sample number
SE	standard error
r^2	square of correlation coefficient
p	probability
c.v.	coefficient of variation
ANOVA	analysis of variance test
t	value of the t-test

CHEMICAL SYMBOLS

H ⁺	hydrogen ion	μgL^{-1}
pH	inverse log of the hydrogen ion concentration	
P	phosphorus	μgL^{-1}
TP	total phosphorus	μgL^{-1}
DP	dissolved phosphorus	μgL^{-1}
N	nitrogen	μgL^{-1}
TN	total nitrogen	μgL^{-1}

TIN	total inorganic nitrogen	μgL^{-1}
TON	total organic nitrogen	μgL^{-1}
IN	inorganic nitrogen	μgL^{-1}
TKN-N	total Kjeldahl nitrogen	μgL^{-1}
NO ₃ -N	nitrite nitrogen	μgL^{-1}
NO ₂ -N	nitrate nitrogen	μgL^{-1}
NH ₄ -N	ammonia nitrogen	μgL^{-1}
Ca	calcium	μeqL^{-1}
Mg	magnesium	μeqL^{-1}
Na	sodium	μeqL^{-1}
K	potassium	μeqL^{-1}
Cl	chloride	μeqL^{-1}
SO ₄	sulphate	μeqL^{-1}
DOC	dissolved organic carbon	mgL^{-1}
DIC	dissolved inorganic carbon	mgL^{-1}
TC	total carbon	mgL^{-1}
Fe	iron	mgm^{-1}
Pb	lead	mgm^{-3}
Mn	manganese	mgm^{-3}
Si	silica	mgL^{-1}
TDS	total dissolved solids	mgL^{-1}
Pb-210	radioactive lead	dpm
SD	secchi depth	m

* M = mass; T = time; L = length

I. INTRODUCTION

To assess current and future proposals for shoreline development of Ontario's inland lakes, scientific tools are needed to evaluate the effects of human use on the lakes and their watersheds. Quantitative rather than qualitative relationships between recreational, as well as non-recreational use and lake and watershed characteristics are required. Useful relationships can often be expressed in the form of predictive mathematical models.

The Lakeshore Capacity Study was initiated to provide such models for lakes in the Precambrian region of southern Ontario. The components of the aquatic and terrestrial systems that were identified as of prime concern to planners and managers responsible for evaluation of development proposals included the wildlife habitat of the terrestrial systems, the fisheries of the aquatic systems, microbially-related human health considerations of the aquatic systems, and the trophic status of the aquatic systems. This report summarizes the research studies carried out by the Limnology and Taxonomy Section (now the Aquatic Ecosystems Section) of the Ontario Ministry of the Environment with the purpose of assessing the effects of shoreline development on the trophic status of lakes.

A. TROPHIC STATUS

The trophic status of a lake is usually assessed by measurement of specific water quality parameters including chlorophyll *a* concentration (a surrogate measurement for phytoplankton biomass), water clarity (often as Secchi disc depth) and rate of loss of oxygen from profundal waters. It is well-known that phosphorus, the algal nutrient present in shortest supply, controls the trophic status of most Precambrian lakes because of its relationship with algal biomass and production. Linkages between total phosphorus content and trophic status formed the basis for the conceptual model developed by Dillon and Rigler (1975) for the assessment of the effects of lakeshore development on the trophic status of lakes. Although Dillon and Rigler (1975) developed predictive models, they were based on the untested assumption that shoreline development using septic systems for waste disposal provides phosphorus to the associated lakes. Many of the models were also developed on the basis of information collected from a large number of lakes located all over the world; lakes in Precambrian regions of Ontario typically exhibit a much smaller range in trophic status. Existing models cannot therefore provide

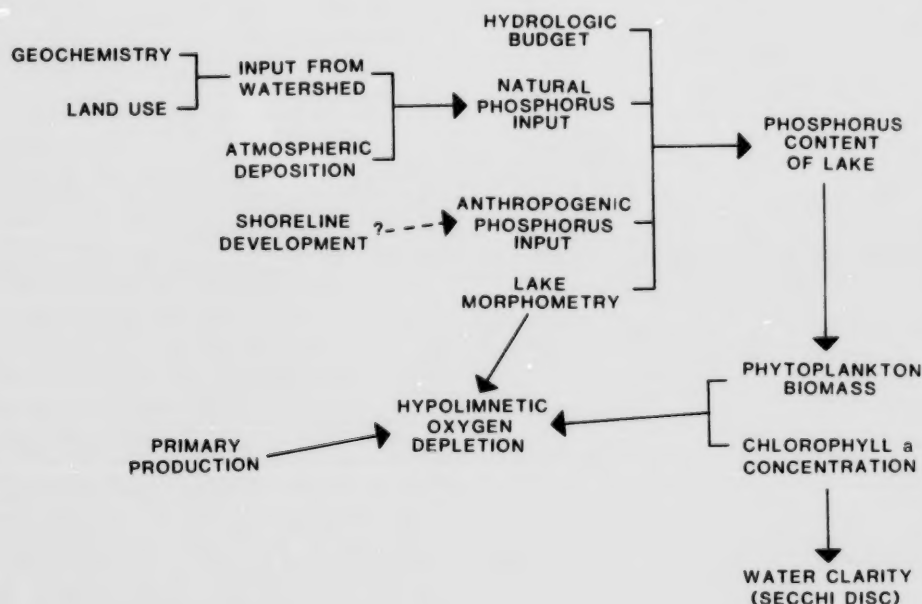


Figure 1 Conceptual model used in the Lakeshore Capacity Study of the relationships linking phosphorus and the trophic status of lakes.

accurate predictions with an acceptable degree of confidence over this smaller range in trophic conditions.

The conceptual model that formed the basis for the trophic status component of the Lakeshore Capacity Study is outlined in Figure 1. The potential effects of shoreline development on trophic status were considered to result from the use of septic systems. Other anthropogenic activities in, on and around lakes were considered to bear little relationship to trophic status.

Other anthropogenic stresses unrelated to shoreline development may affect the water quality of Precambrian lakes. Acidic deposition, a result of the oxidation and hydrolysis of sulphur and nitrogen oxides released as a product of fossil fuel combustion or the smelting of sulphide ores, may be the single most significant environmental problem in eastern North America. Lakes on the Precambrian Shield are particularly sensitive to the deposition of these strong acids. As loading rates of acid in the study area are as high as those in any area in the world, effects on the aquatic biota are to be anticipated (Dillon et al. 1978). In the latter sections of this report, the effects of acidic deposition on trophic status variables are analyzed to ascertain whether nutrient-trophic status relationships are influenced by acidic deposition.

B. STUDY DESIGN

Several approaches were adopted to quantify the effects of lakeshore development on trophic status.

In the first approach, a set of six lakes which, according to existing trophic status models (Dillon and Rigler 1975) exhibited the maximum range in potential significance of shoreline development was selected for study. These six lakes were termed the "A" lakes, and were located in the District Municipality of Muskoka or in Haliburton County.

The potential significance of shoreline development was expressed as the ratio of the estimated potential anthropogenic TP input to the lake (i.e., the potential input from shoreline development) and the estimated total TP input to the lake. These ratios ranged from 0 for Jerry Lake (no shoreline development) to 0.61 for Harp Lake (Table 1). The readily measureable components of the TP mass balances of these lakes were subsequently measured as accurately as possible for four years, as was the hydrologic budget of each lake. Existing trophic status models were calibrated using the subset of the "A" lakes (Jerry and Red Chalk Lakes) with no (or almost no) shoreline development or anthropogenic TP inputs, then applied to the developed lakes. Differences between measured TP concentrations in the developed lakes and levels predicted

Table 1. Estimated ratio of anthropogenic (J_A) to total phosphorus input (J_T) for the "A" lakes.

Lake	J_A/J_T
Harp	0.61
Dickie	0.54
Blue Chalk	0.27
Chub	0.26
Red Chalk	0.04
Jerry	0.0

on the basis of models calibrated using the undeveloped lakes were assumed to be attributable to anthropogenic TP inputs, which in turn, were quantitatively related to the extent of shoreline development on each lake. This approach is described in detail in Section III.

The second approach utilized to assess the effects of lakeshore development on trophic status was to compare the trophic status of selected lakes before and after shoreline or other types of development occurred. No such historical comparison exists for any Precambrian lakes in Ontario. Nine lakes ("B" lakes) were selected. The anticipated changes in development on these lakes are outlined in Table 2. However, few of these changes actually occurred over the course of the Lakeshore Capacity Study. Because of this, monitoring of these lakes by the Ontario Ministry of the Environment has continued (1980-83) and will continue until the proposed developments are carried out or abandoned. Long-term changes in the "B" lakes are not described in this report.

Nevertheless, data collected for the "B" lakes as part of this study were very useful. Information collected for the "A" lakes and their watersheds was used to develop new models linking individual components of the overall conceptual model. Many of these relationships were then tested on the independent data set collected for the "B" lakes. Models were developed for prediction of TP export (or yield) from terrestrial watersheds, for prediction of phytoplankton biomass (and chlorophyll *a* concentration) on the basis of nutrient status, for prediction of water clarity and hypolimnetic oxygen depletion rates, and for other linkages in the conceptual model. Several of these relationships depended on the geology and geochemistry of the terrestrial watersheds of the lakes. Because of this, a set of nine additional terrestrial watersheds was studied to extend the range in geological characteristics beyond that observed for the "A" and "B" lakes. These additional watersheds were termed the "export" watersheds. Model development is described in detail in Section IV.

Table 2. Summary of development status for the "B" lakes. Number of development units at the start of the study, status in 1978, and expected future numbers are indicated.

Lake	Number of development units at start of study (1976)	Number of development units in 1978 (1978)	Expected number of development units
Basshaunt	8	11	31
Glen	5	7	17
Buck	0	0	194
Little Clear	0	0	158
Solitaire	16	14	14
Walker	23	34	73
Bigwind	16	18	100
Crosson	1 ¹	1 ¹	0 ²
Gullfeather	0	0	0 ²

¹ Ontario Ministry of Natural Resources seasonal camp.

² within the boundaries of proposed provincial park and campsite.

II. DESCRIPTION OF THE STUDY AREA AND STUDY LAKES

A. LOCATION

The location of the study lakes, additional ("export") watersheds, and atmospheric deposition stations are indicated in Figure 2. All lakes are located in Haliburton County or in the District Municipality of Muskoka in Ontario. The nearest major urban centres are quite distant from the study area. Toronto and Peterborough are located roughly 200 and 120 km, respectively to the south. North Bay and Sudbury are 130 and 200 km, respectively to the north and northwest.

B. CLIMATE AND VEGETATION

Thirty-year averages of meteorological data taken from the Hydrological Atlas of Canada (1978) indicate that annual precipitation depth averages 90-110 cm in the

Dorset area, the location of the field laboratory in the middle of the study area. A total of 240-300 cm of snow falls each year, generally between December 1 and April 10. January temperatures average -10°C , while in July temperatures average 17.5°C . Mean annual temperature is about 5°C . Lakes in the area generally are frozen from the first week of December to mid-April.

The study area is situated in the Great Lakes-St. Lawrence forest region, a region characterized by eastern white pine, red pine, eastern hemlock and yellow birch. Beech, white oak, sugar maple, basswood, eastern white cedar, red maple, red oak, white birch and poplar are also common in the area. Generally the forest stands on north-facing slopes of the lakes are coniferous, whereas south-facing slopes are deciduous.

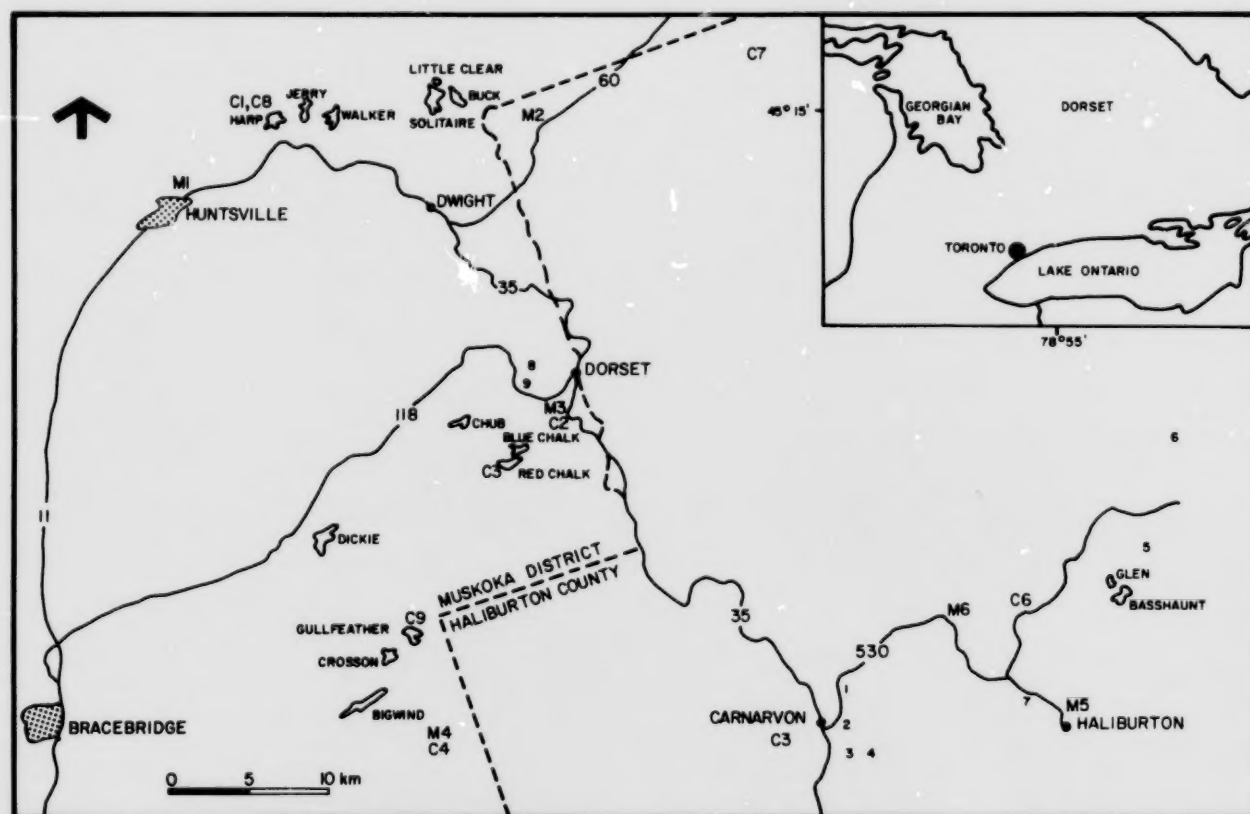


Figure 2 Map of the Lakeshore Capacity Study Area in Muskoka-Haliburton locating the study lakes, meteorological stations (M), bulk precipitation stations (C1 - C9) and export streams (1-Beech 1, 2-Twelve Mile North, 3-Twelve Mile South, 4-Duck 1, 5-Moose 1, 6-Haliburton 12, 7-Head 1, 8-Trading Bay 1, 9-Paint 1).

C. MORPHOMETRY

Detailed bathymetric tables and maps of the Lakeshore Capacity Study lakes are provided in a separate report (Nicolls et al. 1983). A summary of selected morphometric information is provided in Table 3. The lakes have morphometries typical of that for small shield lakes (Brunskill and Schindler 1971). They vary in area from 11 to 111 ha, in mean depth from 5 to 14.2 m and in maximum depth from 12 to 40 m. The ratio of $(A_d + A_o)/V$, a good correlate with lake productivity, ranges from 0.3 to 6.5, again typical of ranges observed for Shield lakes (Schindler 1971).

With the exception of Little Clear Lake, all the lakes have a single, well-developed outflow. Blue Chalk Lake drains into Red Chalk Lake, and Solitaire Lake into Little Clear Lake. Basshaunt, Crosson and Gullfeather Lakes each have small lakes draining into them. All other lakes are headwater lakes with many permanent and ephemeral streams and perhaps small ponds in their watersheds.

D. GEOLOGY

The study lakes and streams are situated on the Canadian Shield, an expanse of Precambrian bedrock covering almost one half of Canada. The Canadian Shield is one of the most seismically inactive areas in the world. The last orogeny (period of mountain building) of the Precambrian period was the Grenville orogeny which formed the Grenville province, a roughly rectangular strip of land 400 km wide and 2,000 km long stretching from Georgian Bay northeast to Labrador and underlying the study area. The bedrock of the Grenville province is of granitic composition, usually granitic gneisses and migmatites with marbles, quartzites, amphibolites and various igneous intrusives (pegmatites, diorites and metabasites) of less importance.

On the scale of individual lake watersheds, bedrock and surficial geology may be variable. As the mineralogy of the bedrock and surficial materials may influence nutrient fluxes, both the surficial and bedrock geology of the study lakes were mapped by air photo analysis and direct field observation. These analyses are interpreted in detail by Jeffries and Snyder (1983) and are summarized below.

Relative contribution to total watershed area of several bedrock and surficial features are listed in Table 4 for the "A" lakes and Table 5 for the "B" lakes. Gneiss sometimes found with thin beds of shist is the dominant bedrock in ten of the study lakes. Migmatite replaces gneiss in five other lakes. The bedrock of Glen Lake is distinct, composed of marble in the northern section contacting gneiss in the southern portion of the watershed.

During the Pleistocene period, four glacial advances, including the most recent or Wisconsin glaciation, have rounded and polished the rock ridges and knobs of the Grenville province, lowering the relief, scouring many river and lake basins and redepositing many types of glacial debris, including moraines, eskers, outwash sands and gravels and lake sediments. The surficial deposits and many characteristics of the watersheds of the lakes are attributable to this glacial activity.

Minor till plains (continuous moraine deposits >1m thick) and thin till deposits (<1 m thick) interrupted by rock ridges are the dominant surficial geological groups. Where thin tills are predominant, such as in Dickie, Crosson and Gullfeather Lakes, peat deposits are generally extensive (Tables 4 and 5). The surficial deposits of Glen Lake are unique among the study lakes. The north end of the watershed has extensive carbonate tills.

The watersheds of Harp, Jerry, Red Chalk and Blue

Table 3. Summary of selected morphometric attributes of study lakes. Parameters are lake area (A_o), volume (V), mean depth ($\bar{z}$), maximum depth (z_{max}), shoreline length (L), development of length (D_L) and volume (D_V) and watershed area excluding A_o (A_d). Notation and methods of calculation after Hutchinson (1957). Details in Nicolls and Reid (1983).

Lake	A_o	V	$\bar{z}$	z_{max}	L	D_L^1	D_V^2	A_d	$\frac{A_d + A_o}{V}$
	(ha)	($m^3 \times 10^5$)	(m)	(m)	(km)			(ha)	(m^{-1})
Basshaunt	47.3	36.6	7.7	24	4.9	1.99	0.96	791	2.29
Bigwind	111.0	118.0	10.7	32	8.2	2.21	1.00	396	0.43
Blue Chalk	49.4	42.1	8.5	23	4.6	1.85	1.11	128	0.42
Buck	40.3	43.9	10.9	30	3.6	1.58	1.09	251	0.66
Chub	32.2	28.5	8.9	27	3.9	1.92	0.99	286	1.12
Crosson	56.8	47.7	8.4	25	3.9	1.45	1.01	501	1.17
Dickie	93.2	46.4	5.0	12	7.8	2.29	1.25	449	1.17
Glen	16.3	11.8	7.2	15	1.8	1.28	1.44	97	0.96
Gullfeather	65.9	31.5	4.8	13	5.3	1.83	1.11	982	3.33
Harp	66.9	82.6	12.4	40	4.6	1.57	0.93	509	0.70
Jerry	50.1	61.9	12.4	35	4.6	1.83	1.06	861	1.47
Little Clear	10.9	8.86	8.1	25	1.5	1.26	0.97	569	6.54
Red Chalk	56.9	80.8	14.2	38	4.8	1.80	1.12	611	0.83
Solitaire	124.0	164.01	13.3	31	6.0	1.51	1.29	379	0.31
Walker	68.2	42.1	6.2	17	6.4	2.20	1.09	258	0.77

¹ Development of length is defined as the ratio of length of shoreline to the length of the circumference of a circle of area equal to that of the lake.

² Development of volume is defined as the ratio of lake volume to that of a cone of basal area A_o and height z_{max} .

Chalk Lakes were partially submerged under glacial Lake Algonquin (about 10,000 years ago) and received sediment laden glacial meltwater. Jerry and Harp Lakes have substantial sand deposits and Red and Blue Chalk Lakes have outwash deposits in consequence (Table 4). Little Clear Lake was intersected by a regional fault which acted as a major glacial meltwater spillway. The largest deposit of sand and gravel is therefore located in the watershed of this lake (Table 5).

E. METHODOLOGY

The methodologies employed in carrying out the trophic status studies are described in detail by Scheider et al. (1983).

Precipitation was collected at up to 9 locations (Figure 2) in the study area between May 1976 and June 1980. Details of collection frequency, periods of collector operation, methods of pooling station data and calculation of elemental deposition are presented by Scheider et al. (1983).

Water samples were collected from all inlet and outlet streams of the study lakes, and from the extra ("export") streams (Figure 2). Stream stage or level was continuously recorded for all "A" lake inflows and outflows and converted to discharge using stage-discharge curves developed for each stage control structure (weir and/or flume). Details of control structure design, stage-discharge relationships, gauging and sampling frequency of the streams,

manipulation of stream data, calculation of discharge from ungauged subwatersheds are provided by Scheider et al. (1983).

Sediment cores were collected from each of the "A" lakes. Methods of collection and chemical analysis are described in detail by Dillon and Evans (1982) and Evans et al. (1983).

All "A" and "B" study lakes were sampled at a single mid-lake station located at the deepest spot in each lake. A station was located at each of the two distinct basins in Red Chalk Lake. Chemical samples were collected at weekly intervals for the "A" lakes and at monthly intervals for the "B" lakes and were volume-weighted composites of all depths when the lake was isothermal. Separate volume-weighted composites were taken from epi-, meta- and hypolimnia when the lakes were stratified. Unlike all other samples, phytoplankton and chlorophyll *a* samples were collected through the euphotic zone, estimated as twice the Secchi disc depth. Lake samples were filtered through 250 μ m mesh to remove coarse particulate material. Details of methods and frequency of sampling and of sample preservation are provided by Scheider et al. (1983).

Analytical methods for chemical analysis of precipitation, lake and stream water and lake sediments are provided by Ontario Ministry of the Environment (1981). Phytoplankton samples were counted by the Utermohl technique.

Table 4. Percent of area of watershed of 6 "A" lakes formed by 4 bedrock types and 7 surficial categories.¹

Lake	Percent of Bedrock Type				Percent of Surficial Type						
	Gneiss	Diorite	Amphibolite and Schist	Migmatite	Minor Till Plain	Thin Till and Rock Ridges	Peat	Bedrock	Outwash	Sand	Pond
Blue Chalk	100	—	—	—	76.7	16.5	—	1.4	5.4	—	—
Chub	100	—	—	—	32.3	61.7	4.4	1.6	—	—	—
Dickie	—	—	—	100	4.7	81.4	13.9	—	—	—	—
Harp	68.5	3.1	28.4	—	49.9	38.5	6.3	0.3	—	4.3	0.8
Jerry	99.5	0.5	—	—	19.8	72.2	4.2	0.1	—	3.4	0.3
Red Chalk	100	—	—	—	57.8	26.5	3.4	2.0	6.9	—	3.6

¹ Categories are described and watershed characteristics analysed by Jeffries and Snyder (1983).

Table 5. Percent of area of watershed of 9 "B" lakes formed by 5 bedrock types and 8 surficial categories.¹

Lake	Percent of Bedrock Type					Percent of Surficial Type							
	Marble	Amphibolite	Gneiss and Schist	Gneiss and Skarn	Migmatite and Granite Intrusives	Till Carbonate	Minor Till Plain	Thin Till and Rock Ridges	Peat Sand	Peat Till	Outwash	Sand	Pond
Basshaunt	—	—	100	—	—	—	56.5	28.4	3.0	1.6	2.9	—	7.6
Bigwind	—	—	—	—	100	—	73.4	21.1	4.4	1.2	—	—	—
Buck	—	—	100	—	—	—	86.6	10.5	—	2.4	—	—	0.5
Crosson	—	0.2	—	—	100	—	20.0	66.4	7.4	—	—	—	6.2
Glen	22.7	—	—	77.3	—	99.9	—	—	—	—	—	0.1	—
Gullfeather	—	0.2	—	—	99.8	—	33.4	50.4	9.3	0.6	—	0.9	5.4
Little Clear	—	4.4	95.6	—	—	—	75.2	—	—	—	24.8	—	—
Solitaire	—	0.3	99.7	—	—	—	81.2	11.4	—	—	6.7	—	—
Walker	—	0.3	99.7	—	—	—	90.3	8.0	—	1.2	—	0.5	—

¹ Categories are described and watershed characteristics discussed in Jeffries and Snyder (1983).

F. LAKE CHEMISTRY

Chemical attributes of the study lakes are summarized in Tables 6 through 9. Excluding Glen Lake, pH, conductivity, major ion and nutrient levels are typical of those observed in dilute, nutrient-poor Canadian Shield lakes (Armstrong and Schindler 1971). The geological setting of the lakes was the major determinant of major ion chemistry and colour of the lakes. Hence, Glen Lake, the only lake with extensive carbonate tills in its drainage basin had much higher alkalinity, pH, conductivity, DIC and TC than all other lakes (Tables 7 and 9). Chub, Dickie, Crosson and Gullfeather Lakes had pH levels of 5.6 – 5.9 while all other lakes had pH >6.2. These four lakes also had higher colour and lower Secchi transparency than all other lakes, probably because the watersheds of these four lakes had extensive deposits of thin till and peat (Table 3).

Average concentrations of total phosphorus (TP) over the four-year period during which data were collected ranged from 5.7 to 12.8 mg m⁻³ among the "A" lakes and 6.4 to 20.7 mg m⁻³ among the "B" lakes (Tables 8 and 9). Levels of total nitrogen (TN = TKN-N + NO₃-N + NO₂-N) ranged from 227 to 394 mg m⁻³ among the "A" lakes and from 296 to 504 mg m⁻³ among the "B" lakes. TN to TP ratios averaged 39:1 (range 28-53) by weight for the "A" lakes and 41:1 (range 24-52) among the "B" lakes. The magnitude of the ratio indicates that phosphorus is the

element present in shortest supply for the primary producers in the lakes (Schindler 1977).

Lakes are often separated into five trophic categories, qualitative expressions of the degree of nutrient enrichment and biological response of the lakes. The categories are ultra-oligotrophic, oligotrophic, mesotrophic, eutrophic and hypertrophic. Often only the three middle categories are employed. The assignment of lakes to particular trophic categories is subjective as clear boundaries between the categories are difficult to identify.

Vollenweider and Kerekes (1980) state that the categorization is not simply an academic exercise despite these limitations as there is a need for limnologists to employ a mutually comprehensible language with each other and with lake managers and planners. Vollenweider and Kerekes (1980) developed a probabilistic approach to trophic designation. They indicate that lakes with TP levels of ≈15 mg m⁻³ have an equal probability (≈45%) of being oligotrophic or mesotrophic. A smaller fraction of such lakes may even be ultra-oligotrophic or eutrophic. Using this equivalence concentration for TP to assign the Lakeshore Capacity Study lakes to trophic categories, Glen Lake (TP = 20.7 mg m⁻³) would be mesotrophic. All other lakes have <15 mg m⁻³ of TP and should be categorized as oligotrophic. Using other trophic state indicators (chlorophyll *a*, Secchi depth) would give slightly different categorizations.

Table 6. Summary of chemistry of the "A" lakes. Results are the mean of 4 annual means, (June 1, 1976 – May 31, 1980) of volume-weighted, composite samples.

Lake	Conductivity (μmho cm ⁻²)	Colour (Hazen Units)	Cations ¹ (μeq L ⁻¹)	Anions ² (μeq L ⁻¹)	pH	Alkalinity ³ (μeq L ⁻¹)
Blue Chalk	30.3	5.2	253	171	6.47	103
Chub	31.5	18.3	250	214	5.58	62.2
Dickie	29.9	15.2	235	207	5.67	58.6
Harp	36.1	6.9	302	215	6.22	101
Jerry	38.3	10.3	328	231	6.19	104
Red Chalk Main	30.6	5.3	260	173	6.24	103
East	33.0	7.9	276	177	6.36	126

¹ Ca²⁺ + Mg²⁺ + Na⁺ + K⁺

² Cl⁻ + SO₄²⁻

³ Corrected fixed-endpoint alkalinity

Table 7. Summary of chemistry of the "B" lakes. Values are the mean of 4 annual means, (June 1, 1976 – May 31, 1980) of volume-weighted, composite samples.

Lake	Conductivity (μmho cm ⁻²)	Colour (Hazen Units)	Cations ¹ (μeq L ⁻¹)	Anions ² (μeq L ⁻¹)	pH	Alkalinity ³ (μeq L ⁻¹)
Basshaunt	46.1	7.0	367	203	6.61	149
Bigwind	31.7	5.8	264	188	6.22	83.4
Buck	33.7	5.5	288	184	6.35	112
Crosson	27.8	12.8	219	187	5.65	58.4
Glen	169.0	5.4	1,740	602	7.23	1,200
Gullfeather	31.1	24.4	255	206	5.84	79.4
Little Clear	39.5	10.7	331	188	6.49	202
Solitaire	32.6	5.5	272	179	6.42	134
Walker	35.7	5.8	303	208	6.43	107

¹ Ca²⁺ + Mg²⁺ + Na⁺ + K⁺

² Cl⁻ + SO₄²⁻

³ Corrected fixed-endpoint alkalinity

Table 8. Summary of the nutrient chemistry of the "A" lakes. Values are the mean of 4 annual means (June 1, 1976 to May 30, 1980). DIC is the mean of two annual values, 1978 to 1980. Samples used to generate means were volume-weighted, composite samples.

Lake	TP	TKN-N (mg m ⁻³)	NO ₃ -N	NH ₄ + -N	DOC	DIC (mg L ⁻¹)	TC	Fe (mg m ⁻³)	Mn	Si (mg L ⁻¹)	Secchi Depth (m)
Blue Chalk	7.07	210	17.5	22.7	3.07	1.03	5.09	80.5	36.7	0.31	6.79
Chub	12.20	296	63.7	33.6	6.19	0.77	8.06	386.0	53.0	1.29	3.29
Dickie	12.80	325	41.1	36.1	6.68	0.59	7.77	242.0	49.3	0.73	2.79
Harp	7.80	245	76.5	13.8	4.80	1.11	6.83	148.0	64.1	1.36	3.90
Jerry	9.14	256	138.0	21.8	5.54	1.26	8.15	237.0	77.1	1.92	3.89
Red Chalk Main	5.71	219	87.7	17.2	3.64	1.25	5.91	128.0	40.6	1.06	6.10
East	7.40	301	43.1	72.3	4.60	1.59	7.81	364.0	64.5	1.25	5.10

Table 9. Summary of the nutrient chemistry of the "B" lakes. Values are the mean of 4 annual means (June 1, 1976 to May 30, 1980) except for DIC which is the mean of two annual values 1978-1980 and TC which is calculated from 1976-1979. Samples used to generate means were volume weighted composite samples.

Lake	TP	TKN-N (mg m ⁻³)	NO ₃ -N	NH ₄ + -N	DOC	DIC (mg L ⁻¹)	TC	Fe (mg m ⁻³)	Mn	Si (mg L ⁻¹)	Secchi Depth (m)
Basshaunt	8.05	263	90.8	26.1	4.47	1.88	7.44	85.8	45.1	1.06	4.68
Bigwind	7.75	237	100.0	19.6	3.83	0.96	5.50	97.3	28.8	0.72	4.99
Buck	7.10	259	66.3	27.8	3.58	1.24	5.88	130.0	63.8	0.65	6.05
Crosson	11.70	289	102.0	28.0	5.16	0.56	6.20	330.0	39.6	0.93	3.66
Glen	20.70	464	60.3	158.0	3.94	14.60	20.40	31.5	59.3	1.97	4.69
Gullfeather	13.40	362	126.0	58.4	6.87	0.77	7.51	462.0	69.8	1.24	2.49
Little Clear	11.20	411	64.4	124.0	4.24	2.01	7.93	605.0	132.0	1.31	5.20
Solitaire	6.40	229	67.0	19.0	2.79	1.17	5.30	48.1	16.5	0.24	7.64
Walker	6.83	309	46.6	34.1	4.56	1.01	6.27	68.2	64.9	0.48	5.55

III. EVALUATION OF THE EFFECTS OF LAKESHORE DEVELOPMENT ON THE TROPHIC STATUS OF THE STUDY LAKES

A. INTRODUCTION

The importance of nutrients, particularly phosphorus, in governing the trophic status of lakes was emphasized in Section I of this report. In this section, we attempt to evaluate the effects of lakeshore development on the trophic status of the study lakes.

The influence of development is very difficult to measure directly because the capability of natural soil deposits around lakes to assimilate or retain phosphorus discharged by septic tile beds is extremely variable. Extrapolations to entire lakes from a few study sites cannot be made with confidence.

Two indirect approaches have therefore been utilized

- a) the nutrient regimes of the developed and undeveloped lakes are compared,
- b) several biological characteristics of the developed and undeveloped lakes that are considered to be dependent on trophic status are compared.

Of the six intensively studied ("A") lakes, two (Jerry and Red Chalk) had very little or no development on their shoreline or in their watersheds, while the other four had significant quantities of development (see Table 1). However, direct comparison of the phosphorus (or other nutrient) content of the developed and undeveloped lakes is not useful for elucidation of the effects of development because of the natural variability in nutrient content in undeveloped lakes. On the other hand, the conceptual links between lakeshore development and a lake's phosphorus budget are well-established. It is therefore possible to formulate the conceptual model in mathematical terms, calibrate it using the undeveloped lakes, then utilize it to ascertain the effects of lakeshore development on the TP budget, and hence, on the trophic status of the four developed lakes.

The second approach employed is the direct comparison of several biological components of the developed and undeveloped lakes. Phytoplankton, zooplankton and macrophytes are considered. However, natural differences in trophic status confound the interpretation of community data, restricting the usefulness of this approach. On the other hand, direct comparison does allow implicit evaluation of the influence of lakeshore development on biological communities that are not related to changes in nutrient supply.

B. THE PHOSPHORUS BUDGET APPROACH

1. BACKGROUND

The factors that determine the total phosphorus (TP) concentration in lakes include lake morphometry, the hydrologic budget, the TP input or loading rate, and the TP loss rate. The conceptual model that links these factors was outlined in Figure 1. The morphometries of the "A" lakes were measured at the beginning of this study. The hydrologic budgets of the "A" lakes were directly measured for the period June 1976-May 1980. The total phosphorus inputs from all significant sources other than those associated with shoreline development were also measured for the same period; these sources include atmospheric deposition and watershed runoff. Loss of TP by outflow from each lake was measured for the duration of the study. Loss of TP by sedimentation within each lake must also be quantified, but the rate of accumulation of TP in the sediments of lakes is very difficult to measure independently. The sedimentation rate of TP can be calculated by difference between the total input rate and the loss rate attributed to outflow if corrections for change in TP concentration in the lake (if any) are made. This methodology is only suitable for the undeveloped lakes since the actual total input is unknown for the developed lakes. Development of a general sedimentation rate model or independent measurement of sedimentation rate is therefore necessary.

The individual components of the hydrologic and TP budgets of the "A" lakes are summarized in the subsequent sections. Because some substances, notably several of the major ions present in freshwaters, behave conservatively with respect to lakes (that is, they are removed from the aquatic system only by outflow but not by biological or chemical processes resulting in sedimentation), their mass balances are employed to validate the hydrologic budgets.

2. HYDROLOGY OF THE "A" LAKES

The components of the hydrologic cycle are related by the water balance equation, an expression of the principle of the conservation of mass. For a lake, this equation can be written as:

$$P + R + G - O - E = \Delta V \quad (1)$$

Where: P = precipitation to the lake surface
 R = surface runoff into the lake
 G = net ground water gain by the lake
 O = outflow from the lake
 E = net evaporation from the lake
 ΔV = change in lake storage, or volume.

Water balances were constructed for four consecutive 12-month periods beginning in June 1976 and for the entire 1976-80 period. The 12-month periods from June to May were chosen as the standard water year for several reasons. Hydrological data collected over the period June 1976 - May 1980 was relatively complete. Mass balance models have been historically used to predict TP concentration in lakes at spring overturn, which commonly occurs in May in the study lakes. Selection of the June - May water year minimizes changes in the water balance resulting from changes in storage since the snowpack is gone and the spring runoff has ended by June 1.

Because each term in the water balance equation except groundwater was measured or estimated independently, it is possible to evaluate the overall accuracy of the hydrologic budgets. This evaluation is performed after each term in the water balance equation is considered.

1) PRECIPITATION

Precipitation depth was measured at six meteorological stations (Figure 2) within the study area. Annual precipitation depth at these locations is summarized in Table 10. Precipitation depth at each lake or calibrated watershed was taken from the nearest station. Mean annual values for the four-year study period ranged from 0.96 m yr⁻¹ to 1.17 m yr⁻¹. Long-term mean annual precipitation depth for the area is 0.9 - 1.1 m yr⁻¹, declining from west to east (Hydrological Atlas of Canada 1978).

There was no significant difference in the annual precipitation depth among the seven stations (using 2-way ANOVA at 0.05 level of significance). Repeating the analysis on a monthly basis, significant differences in precipitation depth among the stations occurred only in February and March. However, on a daily basis considerable variability occurred in some instances. Spatial variability in precipitation depth among five collectors

located on a single lake (Red Chalk Lake) was not significant (Jeffries et al. 1978). Significantly more precipitation fell on the open lake than under the surrounding forest canopy due to interception by the canopy.

The volume of precipitation that fell on each lake and the relative importance of this component in the water balance are summarized in Tables 12-17.

ii) RUNOFF

Annual runoff (annual streamflow/watershed area) for each of the 36 gauged streams is summarized in Table 11. Mean annual values were 0.34 m yr⁻¹; 0.43 m yr⁻¹; 0.54 m yr⁻¹ and 0.56 m yr⁻¹ in 1976-77, 1977-78, 1978-79 and 1979-80 respectively, but watershed-to-watershed variability was quite large. The overall 4-year mean runoff was 0.48 m yr⁻¹, comparing well with the long-term mean annual range in runoff (0.40 - 0.50 m yr⁻¹) in the study area (Hydrological Atlas of Canada 1978). Yield (runoff/precipitation depth) standardizes the runoff data for variations in precipitation depth. Mean annual values of yield for the 36 gauged watersheds were 0.40, 0.49, 0.45 and 0.51 in 1976-77, 1977-78, 1978-79 and 1979-80 respectively.

A detailed discussion of spatial and temporal variability in streamflow will be provided elsewhere as it is beyond the scope of this report. Explanations for the observed differences between watersheds in runoff have also been investigated (Scheider unpub.). Watersheds were grouped based on physical and geochemical characteristics (Jeffries and Snyder 1983), and ANOVA used to assess between-group variance in streamflow indices including annual runoff. Runoff was significantly ($p < 0.05$) greater from watersheds with surficial geology consisting of >3% organic deposits than from those with <3% organic deposits (mean runoff values 0.50 m yr⁻¹ and 0.43 m yr⁻¹ respectively).

Results of stepwise regression using eleven streamflow indices as dependent variables and fifteen watershed indices as independent variables and the remaining ANOVA's on grouped streamflow data will also be described in detail elsewhere.

Table 10. Annual precipitation depth (m) for the study streams and lakes for the 1976-77, 1977-78, 1978-79 and 1979-80 periods.

Station	Lake or Watershed	Annual precipitation depth (m)				Mean
		1976-77	1977-78	1978-79	1979-80	
M1	Harp, Jerry, Walker	0.904	0.906	1.14	0.941	0.973
M2	Buck, Little Clear, Solitaire	0.952	1.23	1.20	1.15	1.13
M3 ¹	Red Chalk, Blue Chalk, Dickie, Chub, Paint Lake #1, Trading Bay #1	0.823	0.818	1.35	1.18	1.04
M4	Bigwind, Gullfeather, Crosson	1.26	0.791	1.07	1.18	1.08
M5	Basshaunt, Glen, Moose #1, Head #1, Haliburton #12	0.809	0.961	1.01	1.04	0.955
M6	12 Mile N., 12 Mile S., Duck #1, Beech #1	1.10	1.14	1.13	1.31	1.17

¹ Precipitation for Dickie Lake was 0.849 m in 1976-77 and 0.806 m in 1977-78.

Table 11. Annual runoff (m yr⁻¹) for 36 gauged streams for 1976-77, 1977-78, 1978-79 and 1979-80 periods.

Watershed		1976-77	1977-78	1978-79	1979-80	Mean
Blue Chalk	1	0.316	0.196	0.231	0.397	0.285
	outlet	0.364	0.403	0.544	0.581	0.473
Chub	1	0.260	0.273	0.363	0.329	0.306
	2	0.371	0.533	0.523	0.626	0.513
	outlet	0.271	0.427	0.515	0.567	0.445
Dickie	5	0.510	0.462	0.552	0.657	0.545
	6	0.388	0.464	0.594	0.569	0.504
	8	0.261	0.322	0.305	0.561	0.362
	10	0.492	0.500	0.638	0.691	0.580
	11	0.298	0.388	0.502	0.502	0.423
	outlet	0.351	0.407	0.496	0.537	0.448
Harp	3	0.467	0.503	0.639	0.636	0.561
	3A	0.262	0.588	0.630	0.530	0.503
	4	0.330	0.411	0.656	0.555	0.488
	5	0.422	0.415	0.553	0.692	0.521
	6	0.201	0.263	0.382	0.324	0.293
	6A	0.275	0.334	0.439	0.458	0.377
Jerry	outlet	0.307	0.440	0.617	0.591	0.489
	1	0.314	0.418	0.527	0.507	0.442
	3	0.400	0.412	0.570	0.549	0.483
	4	0.296	0.343	0.490	0.511	0.410
	outlet	0.394	0.422	0.607	0.490	0.478
Red Chalk	1	0.413	0.443	0.555	0.563	0.494
	2	0.286	0.365	0.532	0.454	0.409
	3	0.373	0.513	0.557	0.540	0.496
	4	0.270	0.415	0.465	0.439	0.397
	outlet	0.372	0.440	0.605	0.594	0.503
Paint	1	0.401	0.529	0.673	0.640	0.561
Trading Bay	1	0.255	0.575	0.786	0.844	0.615
12 Mile North		-	0.499	0.569	0.621	0.563
12 Mile South		-	0.432	0.566	0.529	0.509
Beech	1	0.353	0.451	0.573	0.592	0.492
Duck	1	-	0.521	0.527	0.522	0.523
Head	1	-	0.478	0.540	0.598	0.539
Haliburton	12	-	0.463	0.628	0.559	0.550
Moose	1	0.348	0.501	0.604	0.621	0.519
	Mean	0.343	0.432	0.543	0.555	0.475
n		31	36	36	36	36

Runoff from 12.8% (Red Chalk) to 56.8% (Blue Chalk) of the watersheds of the study lakes was not measured directly.

Estimates of runoff from these ungauged areas were calculated using the mean runoff of the gauged watersheds for each lake (Blue Chalk and Red Chalk Lakes were treated together).

Annual streamflow volume for each of the 6 lake outlets, 21 gauged inlets and the ungauged areas is summarized in Tables A1-A6, with the relative importance of each to the water balance of the lake. These data are used subsequently to calculate nutrient mass balances for each lake.

iii) ENERGY BALANCE AND EVAPORATION

Evaporation was calculated as the residual term in the energy balance equation of each lake:

$$LE = (R - \Delta T)/(1 + B) \quad (2)$$

Where: LE = evaporative heat loss by the lake
R = net radiation

ΔT = change in heat storage by the lake

B = H/LE, H = sensible heat loss by the lake

B may be estimated independently
(Bowen 1926)

Annual lake evaporation rates for the six "A" lakes are summarized in Table 12. Mean annual values were

Table 12. Annual lake evaporation (m yr⁻¹) for 7 study lakes for the 1976-77, 1977-78, 1978-79, and 1979-80 periods.

Lake	Annual Lake Evaporation				
	1976-77	1977-78	1978-79	1979-80	Mean
Blue Chalk	0.759	0.584	0.682	0.601	0.657
Chub	0.808	0.547	0.670	0.666	0.673
Dickie	0.768	0.623	0.731	0.649	0.693
Harp	0.730	0.559	0.676	0.634	0.650
Jerry	0.770	0.570	0.694	0.655	0.672
Red Chalk (main)	0.728	0.566	0.645	0.663	0.651
Red Chalk (east)	-	0.578	0.683	0.657	0.639
Mean	0.761	0.575	0.683	0.646	0.662

0.76 m yr⁻¹, 0.58 m yr⁻¹, 0.68 m yr⁻¹ and 0.65 m yr⁻¹ in 1976-77, 1977-78, 1978-79 and 1979-80 respectively. The 4-year mean value (0.66 m yr⁻¹) compared favourably with the long-term mean annual value of 0.70 m yr⁻¹ for the study area (Hydrological Atlas of Canada 1978).

Differences in annual lake evaporation between lakes were not significant (ANOVA, $p > 0.05$). However, differences between lakes were significant in the months of May, June, July and October, possibly because of the differing heat storage capacities of the lakes. Year-to-year variability in lake evaporation was due to corresponding annual differences in net radiation.

Volumes of water evaporated from the study lakes and the relative importance of this component in the water balance are summarized in Tables A1-A6.

iv) GROUNDWATER

The paucity of surficial material in most of the study watersheds and the impervious nature of the bedrock make it unlikely that groundwater was an important component in the water balance of the study lakes. The intermittent nature of most streams supported this assumption. As other studies of lakes on the Precambrian Shield have demonstrated the relative insignificance of

groundwater in the water balance (Schindler et al. 1976), it was assumed that there was no ground water flux to the lakes.

v) STORAGE

Annual changes in lake storage were estimated by the lake level changes, and were generally small, ranging from -0.23 m yr⁻¹ to +0.16 m yr⁻¹. Only Harp Lake levels were artificially controlled. Lake levels generally declined in 1976-77, rose in 1977-78 and 1978-79, then declined again in 1979-80. Changes in lake level over four years ranged from -0.06 m to +0.15 m.

Changes in lake storage are shown in Tables A1-A6.

vi) WATER BALANCE

Water balances for the four annual periods and from June 1976 to May 1980 are summarized in Tables A1-A6 for Harp, Jerry, Chub, Dickie, Blue Chalk and Red Chalk Lakes, respectively. Surface runoff supplied the greatest amount of water to all lakes, ranging from 89.3% of the total supply in Jerry Lake to 50.6% in Blue Chalk Lake. Precipitation to the lake surface supplied 10.7% - 49.4% of the total supply. Loss of water to the outlet ranged from 72.1% of the total water loss in Blue Chalk Lake to 92.8% in Jerry Lake, the remainder being lost to evaporation.

Table 13. Percentage correction required to balance lake water budgets.

Lake	Component of Water Balance	Year				
		1976-77	1977-78	1978-79	1979-80	1976-80
Blue Chalk	Precipitation	+18.9	+46.0	+ 9.0	+ 20.4	+21.5
	Runoff	+18.3	+38.1	+ 10.2	+ 19.6	+21.0
	Evaporation	-20.5	-64.6	- 17.9	- 40.1	-34.2
	Outlet	-12.0	-26.0	- 6.3	- 11.6	-13.3
	Δ Storage	-68.1	-241	-284	-708	-1,430
Chub	Precipitation	-32.1	+33.9	+ 31.5	+ 46.6	+23.7
	Runoff	- 9.2	+ 7.4	+ 10.5	+ 12.4	+ 6.5
	Evaporation	+32.7	-50.7	- 63.4	- 82.3	-36.7
	Outlet	+ 9.9	- 6.6	- 8.4	- 9.8	- 5.6
	Δ Storage	+564	-369	-721	-590	-16,700
Dickie	Precipitation	+19.3	+34.8	- 11.2	- 32.2	- 2.1
	Runoff	+ 9.2	+13.9	- 6.2	- 13.5	- 1.0
	Evaporation	-21.4	-45.0	+ 20.7	+ 58.5	+ 3.2
	Outlet	- 8.1	-11.9	+ 5.2	+ 12.2	+ 0.8
	Δ Storage	N/A ¹	-249	+758	+246	+400
Harp	Precipitation	-30.9	+45.4	+ 43.1	+ 14.7	+19.6
	Runoff	-10.1	+13.0	+ 11.2	+ 3.1	+ 5.1
	Evaporation	+38.3	-73.5	- 72.8	- 21.9	-29.3
	Outlet	+10.6	-10.9	- 9.3	- 2.7	- 4.5
	Δ Storage	N/A ¹	-280	-2,040	- 136	-1,100
Jerry	Precipitation	+60.5	+48.2	+ 82.5	- 89.8	+25.4
	Runoff	+ 8.3	+ 6.3	+ 9.8	- 9.1	+ 3.0
	Evaporation	-71.0	-76.6	-135.3	+129.0	-36.7
	Outlet	- 7.6	- 5.7	- 8.5	+ 9.5	- 2.8
	Δ Storage	-588	-753	-3,140	+595	-870
Red Chalk	Precipitation	+49.6	+45.2	+ 48.3	+ 56.0	+50.1
	Runoff	+10.5	+ 7.9	+ 11.3	+ 11.4	+10.4
	Evaporation	-56.1	-64.8	- 99.8	- 99.9	-80.0
	Outlet	- 9.3	- 7.1	- 9.2	- 9.5	- 8.9
	Δ Storage	N/A ¹	-324	-2,040	-3,130	-1,670

¹ N/A: storage was not measured.

Annual balances were achieved to within $\pm 10\%$ in all 24 cases (6 lakes $\times$ 4 years) except for Blue Chalk Lake in 1977-78 (-19.7%). A negative balance indicates that loss from the lake exceeded measured supply to the lake. Over the 4-year period, water budgets balanced to within $\pm 10\%$ in all lakes except Blue Chalk (-10.1%), with 4 of 6 lakes balancing to within 5%. Errors in any of the terms of the water balance could potentially account for the imbalance. Because lake storage, precipitation and evaporation are relatively small components in the balance, large errors in these terms ($>100\%$ for storage and $>30\%$ for precipitation and evaporation) would be necessary in most cases to explain the imbalance (Table 13). While relatively small errors ($<15\%$) in terrestrial runoff and loss via the outlet could explain the imbalance in most cases, the bias towards negative balances (19 of 24 cases) may indicate unmeasured groundwater flux into the lakes, especially into Blue Chalk and Red Chalk Lakes. However, the relatively good balances obtained give confidence in the individual supply and loss terms and therefore in the use of the data for the construction of chemical mass balances and for testing the mass balance model.

vii) AREAL WATER LOAD (q_s)

The hydrologic data required for testing or using the chemical mass balance model can be expressed in several ways; as an areal water load ($q_s = Q/A_0$), as a lake water replenishment time ($\tau_w = V/Q$) or as a water replenishment rate or flushing rate ($p = Q/V$), where Q = lake outlet discharge, V = lake volume and A_0 = lake surface area. Since these data are interrelated by the lake morphometry, any one can be considered and used in the mass balance model.

The areal water load (q_s) and the lake water replenishment time (τ_w) for each of the study lakes are summarized in Table 14. Mean values of q_s over the 1976-1980 period ranged from 1.69 m yr^{-1} to 8.69 m yr^{-1} . The coefficient of variation between years ranged from 19% to 29% (mean 23%), so that 95% confidence intervals for q_s were equal to the mean value of $q_s \pm 30\%$ to $\pm 46\%$ for different lakes. There were highly significant differences in q_s between years ($p < 0.00001$; one-way ANOVA with z-scored data) indicating that more than a single year's data are essential for estimating the hydrologic budget of a lake.

3. MAJOR ION MASS BALANCES

A "conservative" substance is defined with respect to a specific system as one which, at steady-state, has an input that is equal to its output; that is, the system does not act as a source or a sink for the substance. For example, Cl^- is usually considered to be a conservative ion with respect to both terrestrial watersheds (Gjessing et al. 1976) and lakes (Sweers 1969). On the other hand Ca^{2+} is generally exported from watersheds in excess of the input from the atmosphere (Likens et al. 1977, Harvey et al. 1981) but behaves conservatively in soft-water lakes (Schindler et al. 1976). In contrast, phosphorus is retained in both lakes (Vollenweider 1968, Vollenweider and Dillon 1974) and in watersheds (Schindler et al. 1976, Dillon and Rigler 1975).

The elements Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- are generally considered to be conservative with respect to lakes; that is, loss to internal sinks by incorporation into the lake's sediments or biota is insignificant relative to the flux or transport rate through the system. Because of this, their mass balances can be used to check the hydrologic budgets of the lakes, since they, in effect, must act like water in the aquatic system. In other words, it is expected that if the water balance is accurately measured, the inputs and outputs of these 5 ions should be equal for any particular lake.

The mean annual concentration of Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- in each of the study lakes is summarized in Table 15. These results indicate that no major changes in ion concentration have occurred over the course of the study; therefore, the concentrations of these ions are in steady-state. The only exception to this is the Cl^- concentration of Harp Lake; a small increase in concentration has been observed each year, with the result that the concentration in the first year is significantly lower than that measured in the fourth year. However, both the Cl^- and K^+ concentrations in all of the study lakes are extremely low. Since these values, which are close to the analytical detection limits, reflect the concentrations of the major inputs (streams, precipitation), the mass balances of these elements are imprecise.

The major ion mass balances for the six "A" lakes are summarized in Tables A7-A12. The total input including that from all streams, the ungauged area of the watershed and the atmosphere is corrected for change in storage in

Table 14. Areal water loads (q_s , m yr^{-1}) and theoretical water replenishment times (τ_w , years) for "A" study lakes for four 12-month periods, mean annual water loads and water replenishment times and coefficient of variation (c.v., %)

Lake	q_s (m yr^{-1})/ τ_w (years)					c.v. (%)
	1976-77	1977-78	1978-79	1979-80	mean	
Blue Chalk	1.30/6.54	1.44/5.90	1.95/4.38	2.09/4.09	1.69/5.23	20
Chub	2.68/3.30	4.22/2.10	5.09/1.74	5.59/1.58	4.40/2.18	29
Dickie	2.04/2.44	2.36/2.11	2.89/1.72	3.12/1.59	2.60/1.97	19
Harp	2.65/4.67	3.78/3.26	5.31/2.33	5.08/2.43	4.20/3.17	29
Jerry	7.17/1.72	7.66/1.61	11.02/1.12	8.90/1.39	8.69/1.46	20
Red Chalk	4.38/3.25	5.17/2.75	7.10/2.00	6.98/2.04	5.91/2.51	23
					23.3	
					mean	

particular. Based on the combined evidence provided by the hydrologic and ion balances, the data collected in the second year is suspect.

The Red Chalk Lake ion and hydrologic balances demonstrated consistent but generally small deficits in input (Tables A6, A12), ranging from 7 to 9% for water and 1 to 18% for the cations.

In almost all cases, particularly for Harp, Chub, Blue Chalk and Red Chalk Lakes, the deficit in the ion budgets is in input. The same trend was measured with respect to the hydrologic budgets, indicating that there is likely a small systematic error in the mass balances of at least these four lakes. There are several possible explanations for these observations. There may be a small input of groundwater (unmeasured) to these lakes. The calculation of the input of water and ions from the ungauged portion of each watershed may be underestimated when calculated by proration of the yields from the gauged watersheds.

In summary, the major ion budgets generally support the contention that the hydrologic budgets were accurately measured. The data collected for Blue Chalk Lake in 1977-78 appear to be the least accurate, and indicate that the TP budget for the same time period is probably poorly measured. The data collected for Chub Lake in 1976-77 and Blue Chalk Lake in 1978-79 and 1979-80 also demonstrated poor ion balances. However, because the hydrologic balances were within 10%, the data collected for these time periods are utilized in the subsequent sections.

4. ATMOSPHERIC DEPOSITION OF NUTRIENTS AND IONS

Atmospheric deposition is a major source of ions to lakes underlain by the sparingly soluble bedrock of the Precambrian Shield. Several investigators have shown that this is particularly true for H^+ , NH_4^+ , and NO_3^- (reviewed by Scheider et al. 1979). Atmospheric deposition may also be a major pathway for introducing nutrients to soft-water lakes. Previous studies of TP budgets for Precambrian Shield Lakes in Ontario, for example, have reported that 5-70% of the TP input to lakes was directly supplied by precipitation to the lake surface (Dillon 1974, Schindler et al. 1976, Cross 1977, Scheider 1978, Scheider et al. 1979).

In this section, the spatial and temporal variability of the atmospheric deposition rates of ions and nutrients, particularly phosphorus, to the study lakes are summarized. Best estimates of annual deposition rates for use in mass balance calculations are generated. Sources of materials deposited from the atmosphere are investigated by examining seasonal trends in deposition and comparing supply rates of materials in "wet-only" (during precipitation events) and "bulk" (wet plus dry) collections.

i) SPATIAL VARIABILITY

Jeffries et al. (1978) found that variations in deposition rates were not significant among five collectors located on a single lake (Red Chalk Lake) for TP, TKN, NO_3^- , H^+ , Ca^{2+} , Mg^{2+} , Na^+ , K^+ , SO_4^{2-} and Cl^- . Significant differences were observed between collectors located on the

lake and those located under the forest canopy adjacent to the lake. Forest canopies certainly modified precipitation chemistry, increasing TP concentrations, for example.

Spatial variability in deposition on a regional scale in the study area was assessed by comparing deposition rates of all nutrients and major ions measured at five of the nine collector locations indicated in Figure 2. These comparisons were made only when all collectors were operating.

Deposition rates of precipitation and all materials measured at all but one of the stations, the most easterly one located in Algonquin Park, were not significantly different ($p > 0.05$ in ANOVA analyses). Therefore, data from all but the Algonquin Park station were pooled to provide an average estimate of deposition applicable to the entire Muskoka/Haliburton area.

ii) TEMPORAL VARIABILITY IN BULK DEPOSITION

Bulk deposition is defined as that collected in a continuously open collector. A summary of the range and median values for monthly bulk deposition (mg or $meq\ m^{-2}\ mo^{-1}$) and corresponding volume-weighted concentrations is given in Table 16. The degree of variation in monthly values reflects the combined variation of precipitation depth and the samples' chemical concentrations. Minimum values in precipitation depth were observed in mid-winter and spring and maximum values in autumn and early winter.

The pattern of temporal variation often indicates the source of materials in the bulk deposition. Seasonal patterns were observed for TP (Figure 3), NH_4^+ , TKN, Ca^{2+} , Mg^{2+} , Na^+ , K^+ and SO_4^{2-} . The deposition of all of these substances was greatest in the warm months of the year. As phosphorus deposited from the atmosphere is

Table 16. Range and median of monthly bulk deposition of ions and nutrients measured at Muskoka-Haliburton from June 1976 to May 1980. Corresponding volume-weighted concentration is also presented. Deposition and concentration are expressed as $meq\ m^{-2}\ month^{-1}$ and $meq\ m^{-3}$ respectively for ions and as $mg\ m^{-2}\ month^{-1}$ and $mg\ m^{-3}$ respectively for nutrients.

Parameter ¹	Monthly Bulk Deposition		Volume-Weighted Concentration	
	Range	Median	Range	Median
H^+	1.00-18.5	5.49	17-136	72
Ca^{2+}	0.37-5.70	1.74	5-102	30
Mg^{2+}	0.12-2.96	0.53	2-35	7
Na^+	0.13-4.31	0.80	1-42	13
K^+	0.02-3.60	0.17	0-43	2
NH_4^+	0.35-7.77	2.29	6-75	30
SO_4^{2-}	0.80-18.0	5.61	19-197	77
Cl^-	0.12-1.32	0.52	3-34	7
NO_3^-	0.81-8.83	2.74	9-89	43
TP	0.35-21.9	1.52	7-472	24
DP	0.26-7.34	0.73	3-140	11
NO_3^- -N	11.4-124	38.4	123-1247	605
NH_4^+ -N	4.92-109	32.1	77-1050	418
TKN	10.4-166	48.3	172-2190	700

¹ Note that NO_3^- and NH_4^+ have been included both as ions ($meq\ m^{-2}\ mo^{-1}$ or $meq\ m^{-3}$) and nutrients ($mg\ m^{-2}\ mo^{-1}$ or $mg\ m^{-3}$).

Table 15. Mean annual concentration of major ions in the study lakes. Sample size varies from ≈ 20 -40; 95% confidence intervals are also shown.

		Ca ²⁺		Mg ²⁺		Na ⁺		K ⁺		Cl ⁻	
Blue Chalk	1976-77	2.82	± 0.10	0.80	± 0.03	0.84	± 0.04	0.38	± 0.02	0.43	± 0.02
	1977-78	2.90	± 0.11	0.79	± 0.02	0.83	± 0.04	0.38	± 0.02	0.47	± 0.02
	1978-79	3.03	± 0.16	0.75	± 0.03	0.83	± 0.05	0.37	± 0.02	0.46	± 0.04
	1979-80	2.98	± 0.10	0.68	± 0.02	0.84	± 0.06	0.39	± 0.02	0.47	± 0.01
Chub	1976-77	2.86	± 0.09	0.75	± 0.02	0.72	± 0.05	0.45	± 0.02	0.56	± 0.03
	1977-78	2.97	± 0.12	0.76	± 0.02	0.79	± 0.04	0.45	± 0.01	0.61	± 0.03
	1978-79	3.07	± 0.12	0.71	± 0.03	0.77	± 0.07	0.42	± 0.01	0.58	± 0.02
	1979-80	2.89	± 0.13	0.64	± 0.02	0.73	± 0.05	0.41	± 0.03	0.60	± 0.06
Dickie	1976-77	2.52	± 0.15	0.64	± 0.04	0.75	± 0.06	0.44	± 0.03	0.79	± 0.06
	1977-78	2.75	± 0.14	0.69	± 0.02	0.85	± 0.05	0.46	± 0.02	0.87	± 0.05
	1978-79	2.82	± 0.10	0.65	± 0.02	0.92	± 0.08	0.43	± 0.01	0.86	± 0.04
	1979-80	2.74	± 0.15	0.58	± 0.03	0.86	± 0.08	0.42	± 0.02	0.87	± 0.07
Harp	1976-77	3.23	± 0.08	1.01	± 0.03	0.94	± 0.05	0.56	± 0.02	0.63	± 0.03
	1977-78	3.21	± 0.10	0.98	± 0.02	0.98	± 0.03	0.56	± 0.02	0.73	± 0.04
	1978-79	3.39	± 0.12	0.96	± 0.04	1.06	± 0.07	0.55	± 0.02	0.76	± 0.03
	1979-80	3.35	± 0.14	0.89	± 0.02	1.03	± 0.06	0.55	± 0.02	0.81	± 0.02
Jerry	1976-77	3.60	± 0.12	1.20	± 0.04	0.86	± 0.07	0.58	± 0.02	0.51	± 0.03
	1977-78	3.55	± 0.09	1.16	± 0.03	0.94	± 0.06	0.56	± 0.02	0.57	± 0.03
	1978-79	3.77	± 0.18	1.17	± 0.06	1.04	± 0.08	0.54	± 0.03	0.54	± 0.04
	1979-80	3.55	± 0.19	1.00	± 0.02	0.90	± 0.08	0.52	± 0.02	0.58	± 0.15
Red Chalk	1976-77	2.80	± 0.08	0.87	± 0.03	0.75	± 0.03	0.44	± 0.01	0.45	± 0.04
	1977-78	2.92	± 0.13	0.85	± 0.02	0.80	± 0.04	0.44	± 0.01	0.46	± 0.02
	1978-79	3.05	± 0.13	0.83	± 0.04	0.82	± 0.07	0.45	± 0.03	0.47	± 0.03
	1979-80	2.92	± 0.14	0.73	± 0.03	0.83	± 0.08	0.43	± 0.01	0.49	± 0.08

the lake as a result of change in lake level. The result is compared to the loss of the ion via outflow from the lake. Minor components of the mass balance are not considered here, but may alter slightly the results discussed subsequently:

- accumulation of each ion in the lakes' sediments is an unmeasured loss,
- input (or output) via groundwater is not considered,
- input from anthropogenic activities (waste disposal systems, road treatment with salts) is not considered,
- small (real) changes in lake concentration may result in changes in storage of each ion in the lakes.

The mass balances calculated for the four-year period for Harp Lake (Table A7) show a consistent deficit of 4-8% (excluding K⁺), comparable to the deficit of 3.6% in the water balance (Table A1). Either a very small systematic error in the hydrologic measurements (overestimation of the outflow volume or underestimation of the inflow volume) has been made or a small unmeasured component of the water balance that contributes ions, e.g. groundwater, may account for the small differences.

The annual mass balances are less precise; variation ranges from about -20 to +20%, either because the accuracy of the measurements is in this range but is averaged out over the longer four-year period, or, more likely, annual changes in water balance are not immediately translated into output because of the finite mixing time and water replenishment time of the lake.

The mass balances obtained from Jerry Lake (Table A8) indicate that the hydrologic balance was accurately measured; Ca²⁺, Mg²⁺ and K⁺ budgets balanced within 3%.

The Na⁺ balance deviated by -7.5%, largely as a result of

a very poor balance (-24.5%) in only one year (1978-79), attributable to very high concentrations of Na⁺ measured in the outflow during spring snowmelt that were observed in no other year.

Excellent balances for all four cations (-0.9% to -3.0%) were obtained for Dickie Lake (Table A10). For both

Jerry and Dickie Lakes, the Cl⁻ balances were poor, at least in part because the concentrations measured in the inflows and the atmospheric deposition were at or near the analytical detection limits. The balances for all elements were also less precise on an annual basis for the same reasons as for Harp Lake.

The Chub Lake major cation balances (Table A9) were acceptable on a four-year basis (+3 to +10%). However, the balances in the first year were very poor; with an excess in input of 42 to 86%. The second, third and fourth-year balances were good, with, in most cases a slight deficit in input. On the other hand, the hydrologic balance was reasonably good in all years (Table A3). A deficit in input was observed in the last three years (-6 to -9%), but a surplus (+7%) was measured in the first year. The difference in hydrologic balance between the first year and the other three years of ~15% was much less than the differences in the ion mass balances.

The Blue Chalk Lake hydrologic balance was the poorest of the study lakes (Table A5), almost certainly because the gauged portion of the lake's watershed was the lowest proportion of the total watershed of any of the lakes. The deficit in input ranged from 5 to 20%, with the second year being by far the poorest. The ion balances also demonstrated a consistent deficiency in input (Table A11), attributable to the last three individual years in

v) DRY VS. WET DEPOSITION

The relative importance of dry vs. wet deposition is reflected in the wet:bulk deposition ratios. Using only data obtained from simultaneous collections of wet and bulk deposition, median values of wet:bulk deposition are summarized in Table 19 for the nutrients and major ions.

Table 19. Ratio of wet deposition to bulk deposition for ions and nutrients (as %). The reported results are the median values of all collections with equivalent collection periods from the Muskoka-Haliburton network from June 1976 – May 1980. The number of sample pairs is indicated (n).

Parameter	Median Percent	n
H ⁺	102	269
Ca ²⁺	73	219
Mg ²⁺	70	222
Na ⁺	100	202
K ⁺	68	219
SO ₄ ²⁻	89	229
Cl ⁻	92	230
TP	47	226
DP	59	160
NO ₃ -N	90	192
NH ₄ ⁺ -N	83	157
TKN	81	195

Wet deposition accounted for 89 – 102% of the bulk deposition for H⁺, Na⁺, SO₄²⁻, Cl⁻ and NO₃. In contrast, dry deposition was an important component of the total deposition for Ca²⁺, Mg²⁺, K⁺, TP, DP, and DOC (wet <75% of bulk). Deposition of the first group of parameters is strongly influenced by the long-range transport of materials from distant sources while the second group is more strongly influenced by the atmospheric suspension of lithologic (soils, dust) and biological (pollen, plant debris) materials from nearby sources.

5. WATERSHEDS AS A SOURCE OF PHOSPHORUS

Phosphorus is transported from terrestrial watersheds to lakes via streamflow. It may be transported in dissolved inorganic or organic forms, as colloidal phosphorus, or in particulate forms ranging from sub-micron size to leaf-litter and other large debris.

The flux of total phosphorus in all forms except the large particulate material has been measured from each of the well-defined subwatersheds of the study ("A") lakes, and from nine additional watersheds (the "export" watersheds) to broaden the range of geological (both bedrock and surficial deposits) types.

Table 20. Areal extent of bedrock and surficial deposit types of the "A" lake subwatersheds and ungauged areas (ung.) expressed as percent. Subwatershed area (A_d) is included. Details in Jeffries and Snyder (1983).

Lake	Sub-watershed	A _d (ha)	Bedrock Type (%)				Surficial Deposit Type (%)						
			Gneiss (Biotited Hornblende)	Diorite	Amphibolite and Schist	Migmatite	Minor Till Plain	Thin Till and Rock Ridges	Peat	Bed- rock	Out- wash	Sand	Pond
Blue	1	27.1	100				94.0	6.0	–	–	–	–	–
Chalk	Ung.	100.4	100				76.3	14.9	–	1.8	7.0	–	–
Chub	1	79.0	100				24.2	72.4	2.8	0.6	–	–	–
	2	121.9	100				16.7	75.3	8.0	–	–	–	–
	Ung.	85.0	100				62.2	32.2	0.9	4.7	–	–	–
Dickie	5	34.4				100	–	74.6	25.4	–	–	–	–
	6	25.4				100	–	78.0	22.0	–	–	–	–
	8	71.4				100	13.7	78.1	8.2	–	–	–	–
	10	71.6				100	–	82.8	17.1	–	–	–	–
	11	109.1				100	–	79.1	20.9	–	–	–	–
	Ung.	136.7				100	10.1	86.6	3.2	0.1	–	–	–
Harp	3	22.6	93		7.0		79.5	11.2	9.3	–	–	–	–
	3A	22.4	42.6		57.4		97.1	–	2.9	–	–	–	–
	4	139.0	13.2		86.8		56.0	32.8	–	0.9	–	7.5	2.7
	5	204.8	100				34.5	48.6	13.3	–	–	3.6	–
	6	21.5	100				45.2	54.8	–	–	–	–	–
	6A	18.9	33.3	66.7			6.6	84.9	8.5	–	–	–	–
	Ung.	79.4	91.5	1.2	7.3		65.8	27.3	1.4	0.3	–	5.5	–
Jerry	1	7.3	100				77.4	22.6	–	–	–	–	–
	3	666.3	100				13.3	78.4	5.2	0.1	–	2.5	0.4
	4	41.0	100				13.6	83.5	–	–	–	2.8	–
	Ung.	146.0	96.4	3.6			44.7	39.1	–	–	–	8.3	–
Red	1	157.1	100				53.0	41.1	–	–	0.8	–	4.9
Chalk	2	32.9	100				–	67.9	10.5	19.4	–	–	2.2
	3	98.7	100				81.7	2.7	9.9	1.2	–	–	4.5
	4	60.3	100				76.2	16.0	2.9	–	–	–	4.8
	Ung.	85.3	100				48.0	17.1	–	1.2	33.7	–	–

primarily of lithologic (soils, dust) or biological (pollen) origin, minimal deposition rates should be anticipated in the winter when the ground is frozen and snow-covered, and the vegetation inactive or snow-covered.

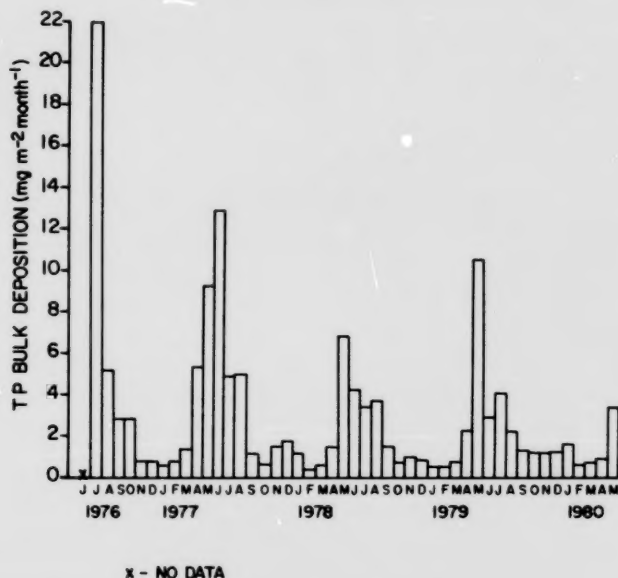


Figure 3 Monthly bulk deposition of total phosphorus (in $\text{mg m}^{-2} \text{mo}^{-1}$) measured at Muskoka-Haliburton from June 1976 to May 1980. Note the seasonal pattern (spring or summer maxima) of deposition.

Many of the seasonal patterns in deposition rate simply reflect the pattern of precipitation depth, with concentration of the substance approximately constant. Certain parameters, particularly TP, also showed elevated concentrations in summer thereby increasing the seasonal nature of their deposition. It should be noted that much of the material deposited by the atmosphere in winter months is stored in the snowpack and released quickly during spring melt. In this respect, the period of greatest input to lakes occurs during the spring months.

iii) PRECIPITATION CHEMISTRY

The median concentrations of TP, other nutrients and ions measured in bulk deposition and wet-only precipitation samples are summarized in Table 17. Concentrations ranged over 1 to 3 orders of magnitude for both sample types depending upon the parameter. Sulphate showed the smallest variability while TP, H^+ , Na^+ , NH_4^+ , DP and NO_3^- exhibited large variations in concentration. On an equivalent basis, the order of importance for major ions in both bulk and wet-only samples was $\text{H}^+ \approx \text{SO}_4^{2-} > \text{NO}_3^- > \text{Ca}^{2+} \approx \text{NH}_4^+ > \text{Na}^+ > \text{Mg}^{2+} \approx \text{Cl}^- > \text{K}^+$. This order of importance is in approximate agreement with that found for other southern Shield locations. Minor differences probably reflect different sampling locations and methodologies.

iv) ANNUAL BULK DEPOSITION

The annual bulk deposition of TP, other nutrients and ions for four annual periods beginning in June 1976 is

Table 17. Range and median concentration of ions ($\mu\text{eq L}^{-1}$) and nutrients (mg m^{-3}) in bulk deposition and wet-only precipitation samples collected at all sites in Muskoka-Haliburton from June 1976 to May 1980.

Parameter ¹	Bulk		Wet-Only	
	Range	Median	Range	Median
H^+	0-457	79	1-1070	76
Ca^{2+}	2-150	29	1-150	24
Mg^{2+}	1-66	6	1-123	4
Na^+	0-174	12	0-113	13
K^+	0-46	2	0-32	1
NH_4^+	0-154	29	2-187	25
SO_4^{2-}	10-323	77	10-252	73
Cl^-	1-138	6	0-113	5
NO_3^-	0-186	36	0-118	31
TP	1-876	20	1-311	10
DP	0-602	8	0-216	4
$\text{NO}_3\text{-N}$	5-2600	500	5-1650	430
$\text{NH}_4\text{-N}$	5-2150	406	20-2620	356
TKN	35-4960	700	65-3360	575

¹ Note that NO_3^- and NH_4^+ have been included under both ions (as meq m^{-3}) and nutrients (mg-N m^{-3}).

summarized in Table 18. Nearly a three-fold variation in annual deposition of TP was observed, indicating the need for long-term precipitation monitoring to obtain reliable average estimates of deposition. Annual variability in the deposition of other chemical species was as great as four-fold over the 4-year period. However, the variation in annual deposition was small compared to between-month variability.

The TP bulk deposition measured in Muskoka-Haliburton fell within the range of values reported by other studies in North America in general, and those located on Shield terrain in particular. Mean deposition over the 4-year study was $37.3 \text{ mg m}^{-2} \text{yr}^{-1}$ while the North American median was $33.9 \text{ mg m}^{-2} \text{yr}^{-1}$. The median North American deposition rate of NO_3^- was $410 \text{ mg m}^{-2} \text{yr}^{-1}$ and of NH_4^+ was $273 \text{ mg m}^{-2} \text{yr}^{-1}$. The corresponding 4-year mean values found at Muskoka-Haliburton (537 for NO_3^- , and 408 for NH_4^+) were both higher than these medians.

Table 18. Annual bulk deposition ($\text{mg m}^{-2} \text{yr}^{-1}$) of ions and nutrients for the composite Muskoka-Haliburton station from June 1976 to May 1980.

Parameter	Bulk Deposition ($\text{mg m}^{-2} \text{yr}^{-1}$)			
	1976-77	1977-78	1978-79	1979-80
TP	58.4	36.5	30.2	21.8
DP	21.5	21.7	13.9	8.69
$\text{NO}_3\text{-N}$	375	499	602	668
$\text{NH}_4\text{-N}$	260	369	552	438
TKN	620	598	993	692
H^+	48.0	63.5	101	84.6
Ca^{2+}	552	608	541	341
Mg^{2+}	106	106	70.1	65.0
Na^+	271	444	314	101
K^+	205	136	95.8	83.3
SO_4^{2-}	3,020	3,360	4,040	3,660
Cl^-	263	296	220	211

Table 21. Areal extent of bedrock and surficial deposit types of the "export" watersheds, expressed as %. Watershed area (A_d) is included. Details in Jeffries and Snyder (1983).

Stream Name	A_d (ha)	Bedrock Type (%)				Surficial Deposit Type (%)							
		Biotite Gneiss (Horn- blende)	Marble	Arkose & Minor Marble	Gneiss, Meta Arkose & Marble	Till (Carbo- nate)	Minor Till Plain	Thin Till and Rock Ridges	Peat	Out- wash	Esker	Drumlin	Pond
Twelve Mile North	426.7	—	11.8	—	88.2	93.0	—	—	2.6	1.4	0.1	—	2.9
Twelve Mile South	171.8	—	2.2	—	97.8	81.0	—	—	8.6	6.7	—	—	3.7
Beech Lake Inflow #1	571.6	—	20.6	—	79.4	89.3	—	—	6.9	—	1.5	0.1	2.2
Haliburton Lake Inflow #12	65.6	100	—	—	—	—	—	—	—	100	—	—	—
Duck Lake Inflow #1	47.3	—	11.9	88.1	—	100	—	—	—	—	—	—	—
Moose Lake Inflow #1	437.9	5.9	94.1	—	—	97.2	—	—	—	—	—	—	2.8
Head Lake Inflow #1	48.3	100	—	—	—	—	13.8	64.2	22.0	—	—	—	—
Paint Lake Inflow #1	21.3	100	—	—	—	—	51.7	38.0	—	5.6	—	—	4.7
Trading Bay Inflow #1	7.9	100	—	—	—	—	55.6	35.5	8.1	—	—	—	—

i) SPATIAL VARIABILITY

Dillon and Kirchner (1975a) have shown that watershed geology influences TP export. Geological characteristics of the 30 individual subwatersheds are summarized in

Table 22. Summary of gross TP export from 21 subwatersheds of the study lakes.

Watershed		TP Export ($\text{mg m}^{-2} \text{yr}^{-1}$)				
		1976-77	1977-78	1978-79	1979-80	Mean
Blue Chalk	1	1.59	1.73	1.06	3.64	2.01
Chub	1	1.97	2.84	2.91	2.67	2.60
	2	12.2	12.9	11.7	17.4	13.6
Dickie	5	18.8	14.1	11.2	17.7	15.5
	6	14.9	23.2	30.5	45.2	28.5
	8	4.67	5.51	3.25	6.95	5.10
	10	21.4	15.5	12.3	14.0	15.8
	11	6.46	8.13	6.68	8.52	7.45
Harp	3	14.6	10.9	14.4	12.6	13.1
	3A	2.10	3.95	3.24	3.30	3.15
	4	5.35	7.48	8.56	7.87	7.32
	5	11.7	8.33	11.7	10.4	10.5
	6	3.47	3.78	4.59	4.07	3.98
	6A	3.61	3.88	4.43	9.76	5.42
Jerry	1	3.95	3.91	5.14	5.18	4.55
	3	10.7	7.88	11.2	8.90	9.67
	4	4.69	2.99	4.70	4.68	4.27
Red Chalk	1	3.97	5.93	6.10	7.99	6.00
	2	3.97	5.14	6.10	5.07	5.07
	3	4.37	7.94	6.52	6.28	6.28
	4	8.82	10.6	8.60	8.18	9.05
Mean		7.78	7.93	8.32	10.0	8.52
Standard deviation		5.79	5.20	6.27	9.12	6.17

Tables 20 and 21. All are small, predominantly forested and 25 of the 30 lie entirely on Precambrian metamorphic bedrock. The other five are underlain in part by sedimentary, carbonate-containing material.

The flux or gross export of total phosphorus, expressed as the yield per unit area of watershed, is summarized in Tables 22 and 23. The mean (1976-80) annual TP export ranged from $2.0 \text{ mg m}^{-2} \text{yr}^{-1}$ (Blue Chalk 1) to $28.5 \text{ mg m}^{-2} \text{yr}^{-1}$ (Dickie 6). Considering individual years, the range was greater ($1.06 - 45.2 \text{ mg m}^{-2} \text{yr}^{-1}$).

Table 23. Gross TP export from 9 "export" watersheds.

Watershed	TP Export ($\text{mg m}^{-2} \text{yr}^{-1}$)				
	1976-77	1977-78	1978-79	1979-80	Mean
Paint #1	2.02	2.33	2.12	2.05	2.13
Trading Bay #1	2.56	3.52	3.84	6.23	4.04
Beech #1	5.86	5.84	6.60	11.2	7.38
Duck #1	—	3.19	2.95	4.43	3.52
12 Mile N	—	6.67	7.61	10.5	8.26
12 Mile S	—	6.34	7.55	8.37	7.42
Haliburton #12	—	6.92	9.47	10.8	9.06
Head #1	—	6.51	10.6	16.6	11.2
Moose #1	5.55	6.94	6.91	7.95	6.83
Mean (metamorphic watersheds)	2.29	4.82	6.51	8.92	6.61
Standard deviation	0.38	2.25	4.16	6.24	4.23
Mean (sedimentary watersheds)	5.70	5.80	6.32	8.49	6.68
Standard deviation	0.22	1.51	1.93	2.65	1.84

Two-way ANOVA was used to evaluate the temporal and between-site variability in TP export. There were no significant differences between years ($p > 0.10$). Differences between individual watersheds and between groups of watersheds (all inlets to each lake forming one group) were significant ($p < 0.01$, $p < 0.001$, respectively). The mean TP export for each watershed group, summarized in Table 24, ranged from 2.0 for Blue Chalk Lake to 14.5 for Dickie Lake, with the other 4 groups averaging 6.2 – 8.1 $\text{mg m}^{-2} \text{yr}^{-1}$.

There was no significant difference in TP export between the metamorphic watersheds (21 of the subwatersheds of the study lakes plus 4 of the "export" watersheds) and those which have carbonate-containing sedimentary material in their watersheds (5 "export" watersheds). Bedrock type alone was not related to gross export of TP for the study watersheds, in contrast with the suggestion of Dillon and Kirchner (1975a).

Although year-to-year variability was not significant compared to site-to-site variability, short-term temporal variability was evident. For example, the monthly TP

export for each of the Harp Lake subwatersheds (Figure 4) varied by 1-2 orders of magnitude within each subwatershed. The greatest exports occurred during the periods of high streamflow, particularly the spring snow-melt period.

Table 24. TP export from each of the "A" lakes' watershed. Results are the areally weighted averages of all subwatersheds of each lake.

Watershed Group	TP Export ($\text{mg m}^{-2} \text{yr}^{-1}$)				Group Mean
	1976-77	1977-78	1978-79	1979-80	
Blue Chalk	1.59	1.73	1.06	3.64	2.01
Chub	7.09	7.90	7.31	10.0	8.08
Dickie	13.2	13.3	12.8	18.5	14.5
Harp	6.81	6.39	7.82	8.00	7.25
Jerry	6.45	4.93	7.01	6.25	6.16
Red Chalk	5.28	7.40	6.83	6.88	6.60
Year Mean	7.78	7.93	8.32	10.0	

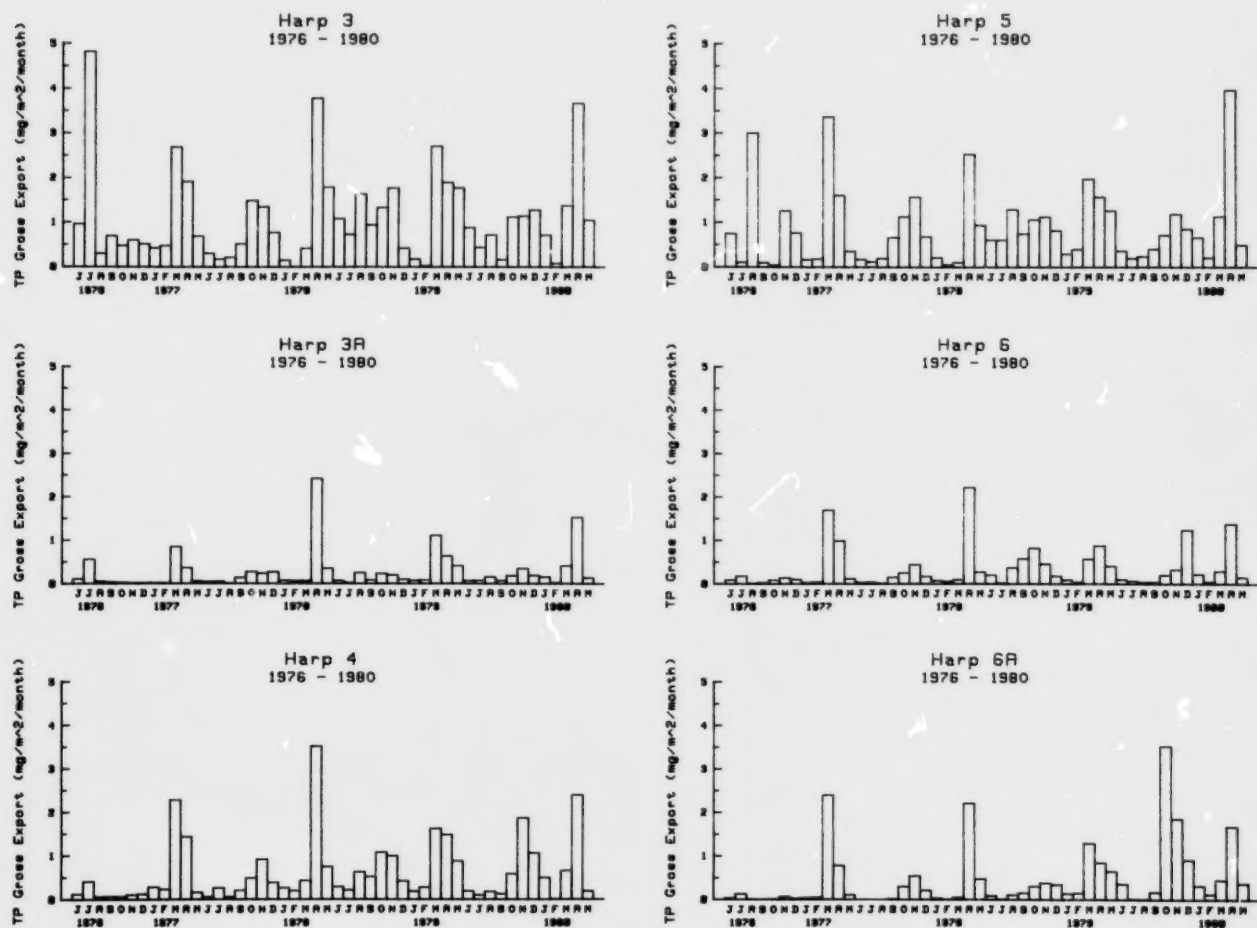


Figure 4 Monthly gross export of total phosphorus (in $\text{mg m}^{-2} \text{mo}^{-1}$) measured for six Harp Lake subwatersheds from June 1976 to May 1980.

ii) WATERSHED CHARACTERISTICS AND TP EXPORT

The earliest attempt to classify watersheds on the basis of the total phosphorus export was made by Vollenweider (1968) who noted that watersheds on "unproductive primitive rock" with corresponding "oligotrophic" soils usually had a total phosphorus export of $< 20 \text{ mg m}^{-2} \text{ yr}^{-1}$, while more fertile watersheds, which were often used for agriculture, had TP exports of $> 20 \text{ mg m}^{-2} \text{ yr}^{-1}$. Dillon and Kirchner (1975a) subsequently categorized all watersheds for which TP export was reported according to land use and to bedrock geology. Major land use categories considered were forest, pasture (defined as cleared but uncultivated land), agriculture and urban. Bedrock was separated only into sedimentary and igneous groupings although the latter classification inadvertently included metamorphic watersheds. When only watersheds that were completely forested or had a combination of forest and cleared land with no agriculture were considered, it was found that the TP export could be estimated based on these two land uses and the bedrock geology (Table 25). Watersheds that were forested and on igneous bedrock had the lowest TP export (average of $4.7 \text{ mg m}^{-2} \text{ yr}^{-1}$; range of $0.7\text{--}8.8 \text{ mg m}^{-2} \text{ yr}^{-1}$), while sedimentary watersheds had higher export if totally forested (mean of $10.7 \text{ mg m}^{-2} \text{ yr}^{-1}$) or if cleared land was also present (mean of $23.3 \text{ mg m}^{-2} \text{ yr}^{-1}$).

Table 25. Ranges and mean values for export of total phosphorus from 39 watersheds. Results are in $\text{mg m}^{-2} \text{ yr}^{-1}$ (Dillon and Kirchner 1974).

Land Use	Geological Classification	
	Igneous	Sedimentary
Forest		
Range	0.7-8.8	6.7-14.5 ¹
Mean	4.7	10.7 ¹
Forest + Pasture ²		
Range	5.9-16.0	11.1-37.0
Mean	10.2	23.3

¹ Values corrected from original source.

² 15% or more of the watershed cleared.

Watersheds whose land use could be classified at least in part as "agriculture" or "urban" had much higher losses of phosphorus, reaching in some cases $1000\text{--}2000 \text{ mg m}^{-2} \text{ yr}^{-1}$. Dillon (unpublished manuscript) subsequently evaluated more recent information using a similar approach. An approximate doubling of the number of watersheds for which total phosphorus export was measured resulted in virtually no change in the TP export-land use-geology relationship (Table 26).

Kirchner (1975) attempted to relate the TP export from one land use-geology class of watersheds (forested, igneous) to physical characteristics of the watersheds. He found that only drainage density (D_d , the total stream length per unit area of watershed, km^{-1}) was significantly correlated to TP export for these watersheds:

$$\text{Export (TP)} = 1.14 + 5.85 D_d; (r^2 = 0.88, n = 18) \quad (3)$$

Such a relationship can provide the means for more accurate prediction of the TP export for specific watersheds or subwatersheds within the general classification scheme.

Numerous additional watershed studies have been carried out since the work of Dillon and Kirchner. The total phosphorus export from 928 watersheds in the United States was measured as part of the United States National

Eutrophication Survey for a 1-year period. Of these, 69 were predominantly ($\geq 90\%$) forested. The mean TP export from these 69 watersheds was $9.1 \text{ mg m}^{-2} \text{ yr}^{-1}$, but no distinction was made on the basis of geology (Omernik 1977). Both sedimentary and igneous watersheds were included.

A study of 23 small drainage areas in Finland for a 10-year period (Kauppi 1979) showed that $4.0\text{--}5.6 \text{ mg m}^{-2} \text{ yr}^{-1}$ were lost from forested watersheds with no human habitation. Many studies (e.g., Jones et al. 1976) have included watersheds where land use was at least in part agricultural or urban. The exports are always higher, often much higher, than in cases where forest predominates. In a number of instances, TP export has been found to be statistically related to anthropogenic activities, e.g., fertilizer usage rates, human or animal population density, etc. (Omernik 1977). However, no new predictive relationships for TP export have been developed for forested watersheds.

Table 26. Ranges and mean values for export of total phosphorus from 69 watersheds. Results are in $\text{mg m}^{-2} \text{ yr}^{-1}$ (Dillon, unpublished manuscript).

Land Use	Geological Classification	
	Igneous	Sedimentary
Forest		
Range	0.7-12.2	6.7-14.5
Mean	5.5	10.3
Forest Pasture ¹		
Range	5.9-16.0	5.8-37.0
Mean	9.8	19.8

¹ 15% or more of the watershed cleared.

The TP export measured for the watersheds of the study lakes and the "export" watersheds have a greater range than previously reported for Precambrian watersheds. However, the mean TP export values for the four years lie within the range summarized in Table 26 in 20 out of 25 watersheds on metamorphic bedrock, with five greater than previously reported, and in four out of five watersheds partially underlain by sedimentary rock, the one measured export (Duck 1) being lower than previously reported. The grand mean TP export of all the metamorphic basins ($8.2 \text{ mg m}^{-2} \text{ yr}^{-1}$) was higher than the mean ($5.5 \text{ mg m}^{-2} \text{ yr}^{-1}$) reported in Table 26 based on previous studies.

6. ANTHROPOGENIC SOURCES OF TOTAL PHOSPHORUS

Human activities may contribute phosphorus (TP) to a lake in many ways. The discharge of TP to a lake from sewage disposal systems, the direct application of TP to

the terrestrial watershed in the form of lawn or garden fertilizers, the alteration of TP supplied to the lake from the watershed because of changes in the land use patterns (e.g., clearing land, constructing roads), the input of TP to the lake by the addition or removal of fill and the mixing of sediment TP into the overlying water through agitation with motorboats are examples of these activities. Human activities must be evaluated on a lake-by-lake basis to estimate anthropogenic TP supply. Phosphorus input from waste disposal systems was considered to be the most important potential source of anthropogenic TP supplied to the study lakes.

Several factors influence the supply of TP to lakes from waste disposal systems; namely, i) the input of TP to the sewage disposal system, ii) the removal or retention of TP by the sewage disposal system, and iii) the transport and retention of TP by the soil and biota between the sewage disposal system and the lake.

i) INPUT OF TP TO SEWAGE DISPOSAL SYSTEMS

The TP supplied to the sewage disposal systems was estimated in three ways. First, a mean value of TP concentration in septic tank effluents was calculated (13.2 mg L^{-1}) based on 21 values in the literature (Table 27). Assuming effluent TP concentration approximates influent concentration, the product of TP concentration and the average per capita water use (164 L/person/day , based on 10 literature values summarized in Table 28) allowed an estimate of $0.79 \text{ kg C}^{-1}\text{a}^{-1}$ (kilograms per capita per annum). This value agrees closely with a commonly used earlier estimate of $0.80 \text{ kg C}^{-1}\text{a}^{-1}$ based on 13 studies in Europe and North America (Vollenweider 1968). In the second method, data on TP supply to

Table 27. Concentration of total phosphorus ([TP] in mg L^{-1}) in septic tank effluents listed in 21 studies.

[TP]	Source
5	Hansel (1968)
6.5	Preul (1965)
6.9	Ali (1980)
7.7	Barshied and Baroudi (1974)
8.2	Lake George Water Research Centre (1971)
9.8 ¹	Ali (1980)
10.0	Feth (1966)
11.6	Viranaghavan and Warnock (1974)
11.8	Brandes (1972)
12.1	Chan (1978)
13.3	Bouma et al. (1972)
14.2	Brandes (1978)
14.6	Otis et al. (1977)
15	Brandes (1977)
15	Brandes (1972)
15.3	Otis et al. (1975)
18 ²	Bennett and Linstedt (1975)
19.6	Brandes (1976)
20	Schroepfer and Preul (1964)
20 ³	University of Wisconsin (1973)
21.8	Harkin et al. (1979)
13.2	mean value ($n = 21$)

¹ Raw sewage at Toronto's sewage treatment plants.

² Average for U.S.A.

³ Average for Wisconsin.

sewage treatment plants in Ontario (Appleby 1978) and the U.S.A. (Wall and Webber 1970) allowed estimates of $1.4 \text{ kg C}^{-1}\text{a}^{-1}$ and $1.36 - 1.59 \text{ kg C}^{-1}\text{a}^{-1}$ respectively. Finally, Bucksteeg (1966) estimated $0.55 \text{ kg C}^{-1}\text{a}^{-1}$ as the amount of TP ingested in food; presumably a comparable quantity was excreted. The value of $0.55 \text{ kg C}^{-1}\text{a}^{-1}$ is probably too low as an estimate of TP input to waste disposal systems because it ignores TP in non-food items such as cleaners and detergents. On the other hand, the sewage treatment plant data are probably high because they include TP from human wastes (43% total), industrial and domestic cleaning compounds (29%) and from miscellaneous sources (28%, Appleby 1978).

Table 28. Measured household¹ water use data for year-round dwellings.

Water Usage (L/person/day)	Source
118	Brandes (1976)
151	Otis et al. (1976)
156	Laak (1975)
159	Witt et al. (1974)
161	Siegrist et al. (1976)
168	Chowdry (1974)
173	Chan (1978)
180	Ligman et al. (1974)
184	Harkin et al. (1979)
186	Chan (1978)
164	mean value ($n = 10$)

¹ Ontario Dept. of Health (1969) estimated 55 L/person/day as usage for cottages. Chan (1978) estimated (no data) 91 L/person/day for cottages. Methods for estimation were not specified.

A value of $0.60 \text{ kg C}^{-1}\text{a}^{-1}$ can be calculated as the P in human waste from sewage treatment plant data. The value chosen as the TP supply to sewage disposal systems is $0.80 \text{ kg C}^{-1}\text{a}^{-1}$.

ii) RETENTION OF TP IN SEWAGE DISPOSAL SYSTEMS

The retention of TP by the sewage disposal system varies with the type of system. The four types of systems commonly used in the study area were the septic tank with tile field, the holding tank, the leaching pit and the aerobic system. In the case of a holding tank, all waste is removed from the watershed and, consequently, input of TP to the lake is zero. No TP is removed in the leaching pit or the aerobic system if the compost is eventually emptied within the watershed.

The septic tank and tile field was the most common type of system in use. Sludge accumulation in the septic tank may remove 5% (Brandes 1976) to 20-30% (Hardisty 1974) of the TP in the waste. Further TP removal in the tile field and between the tile field and the lake is dependent upon soil characteristics such as soil texture (proportion sand, silt, clay), soil drainage (frequency of saturation), soil structure (the arrangement of the primary soil particles such as sand and clay into aggregates) and slope (Long 1979) as well as environmental factors such as oxygen, temperature and climatic condi-

tions. Estimates of TP retention in soils in Ontario range from 1% (Brandes 1974) to 99% (Chan 1978). In general, fine-grained soils with high clay content are more effective in the attenuation of TP than are sandy soils or soils with high gravel content. Phosphorus retention coefficients of septic tile fields of differing characteristics are given in Table 29.

The removal of TP by septic systems depends upon their proper maintenance and design. A summary of literature data on the failure rate of septic systems in Ontario (Table 30) showed that an average of 61% were not designed or constructed properly, or maintained in a satisfactory condition.

On the basis of this information, a conservative approach in calculating the potential TP input from anthropogenic

sources has been taken and a value of $0.80 \text{ kg C}^{-1}\text{a}^{-1}$ for septic tank/tile field systems, leaching pits and aerobic systems has been assumed. If it is known that the septic system is in satisfactory condition and information on the soil type is available, then a retention factor may be applied using values outlined in Table 29.

iii) HUMAN USAGE OF STUDY LAKES

The quantification of human usage of the study lakes was carried out by the Land Use Component, Lakeshore Capacity Study. Results on the number of units, type of sewage disposal facilities and cottage use statistics for the lakes are summarized in Table 31.

No data were available for Jerry Lake, but very little use is made of the one dwelling on this lake. Use was estimated as 0 capita years yr^{-1} .

Human usage of Red Chalk Lake was calculated as the mean value of usage estimated by three methods, accounting for the observation that two of the three dwellings were mostly vacant during the 1976-1980 period. The 'accessibility equation' (Downing 1983) and the 'cottager equation' (Downing 1983) gave estimates of $0.84 \text{ cap yrs yr}^{-1}$ and $0.77 \text{ cap yrs yr}^{-1}$ respectively. Extrapolation of usage data for Blue Chalk Lake allowed an estimate of $0.64 \text{ cap yrs yr}^{-1}$ as the usage of Red Chalk Lake. The mean value of the three estimates was $0.75 \text{ cap yrs yr}^{-1}$. It was assumed that the dwellings did not utilize holding tanks.

iv) ESTIMATION OF POTENTIAL ANTHROPOGENIC TP INPUTS

The potential anthropogenic TP input for each lake was calculated using the following equation:

$$\text{TP Input (kg yr}^{-1}\text{)} = 0.80 \text{ kg TP C}^{-1}\text{a}^{-1} \cdot \text{usage (cap yrs yr}^{-1}\text{)} \cdot [1 - \text{fraction of dwellings on holding tanks}] \quad (4)$$

Table 29. The retention coefficients (R_s) of total phosphorus for septic tile filter beds of different characteristics. Results are based on 4 years of data (from Brandes et al. 1974). D_{10} is grain size.

Filter Bed	R_s
1. 22 in. sand ($D_{10} = 0.24 \text{ mm}$) 8 in. mixture 4% red mud, 96% sand	0.76
2. 30 in. sand ($D_{10} = 0.30 \text{ mm}$)	0.34
3. 30 in. sand ($D_{10} = 0.60 \text{ mm}$)	0.22
4. 30 in. sand ($D_{10} = 0.24 \text{ mm}$)	0.48
5. 30 in. sand ($D_{10} = 1.0 \text{ mm}$)	0.01
6. 30 in. sand ($D_{10} = 2.5 \text{ mm}$)	0.04
7. 15 in. sand ($D_{10} = 0.24 \text{ mm}$) 15 in. mixture 10% red mud, 90% sand	0.88
8. 15 in. sand ($D_{10} = 0.24 \text{ mm}$) 15 in. mixture 50% limestone, 50% sand	0.73
9. 30 in. silty sand	0.63
10. 15 in. sand ($D_{10} = 0.24 \text{ mm}$) 15 in. mixture 50% clay-silt, 50% sand	0.74

Table 30. Summary of cottage pollution survey results obtained by the Ministry of the Environment (from Chan 1978).

Classification ¹	17 Lakes (1971)	29 Lakes (1972)	Shebandowan Lakes (1976)	Muskoka Dickie Lake (1976)	Peterborough District (1975)	Peterborough Victoria (1974)	Muskoka Bass Lake (1974)	Muskoka Bala Lake (1975)
Satisfactory	66	39	37	25	20	61	32	36
Direct Polluter	3	2	2	3	7	—	4	1
Substandard	18 ²	31 ²	40	26	45	39	17	21
Nuisance	13	28	19	39	26	—	38	35
Unclassified	—	—	2	7	2	—	9	7

¹DEFINITIONS:

Satisfactory — A system which meets all current standards of good design, construction and location, and is properly maintained.

Direct Polluter — A system which permits sewage to contaminate the groundwater, or to reach the lake either by direct discharge through a pipe or ditch or over the ground surface.

Substandard — A system which does not meet current standards of design, construction and location, and/or is in a state of neglect.

Nuisance (Wash Water) — A system causing wash water to be exposed on the surface of the ground either directly through a waste pipe, escaping from a seepage pit or just thrown on ground surface.

Nuisance (Toilet or Solid waste) — A system causing a waste containing fecal or urinary discharges to be exposed on the surface of the ground, either directly through a pipe or escaping from some part of sewage disposal system including a privy.

² % of substandard and unclassified systems combined.

Table 31. Results of survey of numbers of occupied dwelling units (No. Units), usage of the units (capita yr⁻¹) and sewage disposal types (%) – septic tank and tile field (1), leaching pit (2), holding tank (3) or aerobic system (4). NI indicates information is not available.

Lake	No. Units			Sewage Disposal Type				Usage		
	1976	1977	1978	1	2	3	4	1976	1977	1978
Basshaunt ⁴	8	11	11	83	–	–	17	6.73	9.25	9.25
Bigwind ⁵	16	–	18	80	20	–	–	5.98	–	6.72
Blue Chalk ⁴	11	–	11	80	20	–	–	7.07	–	7.07
Buck	0	–	0	–	–	–	–	0	–	0
Chub ⁵	10	–	11	60	20	20	–	4.01	–	4.41
Crosson ⁶	1 ³	1 ³	1 ³	83	–	17	–	–	4.84	–
Dickie ⁵	119	–	125	77	10	13	–	108.2	–	113.6
Glen ⁴	5	7	7	100	–	–	–	5.59	7.83	7.83
Gullfeather	0	–	0	–	–	–	–	0	–	0
Harp ⁵	80	–	83	100	–	–	–	99.8	–	103.5
Jerry	1	–	1	NI	NI	NI	NI	0 ¹	–	0 ¹
Little Clear	0	–	0	–	–	–	–	0	–	0
Red Chalk	3	–	3	NI	NI	NI	NI	0.75 ²	–	0.75 ²
Solitaire ⁴	16	–	14	75	25	–	–	16.7	–	14.6
Walker ⁵	23	29	34	100	–	–	–	51.0	64.3	75.4

¹ Estimated, the cottage was very rarely in use.

² Estimated, see text.

³ Ontario Ministry of Natural Resources seasonal camp.

⁴ Data on sewage disposal type based on 1976 survey.

⁵ Data on sewage disposal type based on 1978 survey.

⁶ Data on sewage disposal type based on information from 1977.

A summary of the potential TP inputs for the study lakes is given in Table 32. The lag time between the input of TP to the sewage disposal system and the input of sewage TP to the lake was assumed to be zero. Again, this is a "worst-case" assumption, especially in the case of newly constructed dwellings (e.g., Walker Lake).

Development indices may more appropriately reflect the potential effect of human use on lakes if corrected for lake morphometry. Indices calculated per unit lake area and volume are indicated in Table 33. The ranking of lakes with respect to degree of development does change slightly depending on the index selected. However, of the "A" lakes, Harp and Dickie, have the greatest quantity of development based on all indices, Chub and Blue Chalk an intermediate amount, and Jerry and Red Chalk little or none.

7. MINOR COMPONENTS OF THE TOTAL PHOSPHORUS BUDGETS

In addition to the major components of the phosphorus budgets described individually in the preceding sections, several other inputs and outputs of TP to and from the study lakes may make less significant contributions to the TP budgets.

The harvest of sport fish by anglers represents a loss of TP from the study lakes which may in part be balanced by fish stocking. TP is lost from lakes in the form of emerging insects. It is reintroduced in leaf fall, mainly in the autumn. Nesting birds, if they aggregate in large colonies, may make a substantial contribution to the nutrient budget of lakes.

While none of these pathways of TP was assessed directly in the study lakes, it is possible to estimate the magnitude

Table 32. Potential anthropogenic TP supply to the study lakes from lakeshore waste disposal systems.

Lake	Potential TP Supply (kg TP yr ⁻¹)		
	1976	1977	1978
Basshaunt	5.38	7.40	7.40
Bigwind	4.78	–	5.38
Blue Chalk	5.66	–	5.66
Buck	0	–	0
Chub	2.57	–	2.82
Crosson	–	3.21	–
Dickie	75.3	–	79.1
Glen	4.47	6.26	6.26
Gullfeather	0	–	0
Harp	79.8	–	82.8
Jerry	0	–	0
Little Clear	0	–	0
Red Chalk	0.60	–	0.60
Solitaire	13.4	–	11.7
Walker	40.8	51.4	60.3

Table 33. Indices of development for the study lakes. Use measured in capita-years year⁻¹ (cap-yr yr⁻¹).

Lake	Average Usage		
	Total (cap yr yr ⁻¹)	Areal (cap-yr ha ⁻¹ .yr ⁻¹)	Volumetric (cap-yr 10 ⁶ m ⁻³ .yr ⁻¹)
Basshaunt	8.4	0.18	0.023
Bigwind	6.4	0.058	0.005
Blue Chalk	7.1	0.14	0.016
Buck	0	0	0
Chub	4.2	0.13	0.015
Crosson	4.8	0.085	0.010
Dickie	111	1.19	0.238
Glen	7.1	0.44	0.061
Gullfeather	0	0	0
Harp	102	1.52	0.123
Jerry	0	0	0
Little Clear	0	0	0
Red Chalk	0.75	0.013	0.001
Solitaire	15.7	0.13	0.010
Walker	63.6	0.93	0.150

of these additional fluxes. In this section loss of TP in emerging Diptera and in fish harvest corrected for fish stocking is estimated for Harp Lake. Additionally, the significance of leaf litter as a source of TP is assessed for the A lakes.

i) ANGLING AND FISH STOCKING AS TP PATHWAYS FOR HARP LAKE

The quantity of TP removed each year in harvests of sport fish from Harp Lake was estimated as the product of predicted annual harvest (H'' , in $\text{kg ha}^{-1} \text{yr}^{-1}$), lake area (A_0 , in ha) and TP concentration of whole fish. H'' was estimated by solving the sequential series of regression equations developed by McCombie (1982) in the Fisheries Component Lakeshore Capacity Study. Firstly, summer fishing effort by cottagers (E_c) was calculated given $A_0 = 66.9$ ha for Harp Lake, and N (number of dwellings) was 83, as below:

$$\ln E_c = 1.832 + 1.664 \ln A_0 - 0.119 (\ln A_0)^2 + 0.204 \ln N \quad (5)$$

E_c was 2049 angler hrs. E_c/A_0 was 30.6 angler hrs ha^{-1} . Summer effort by non-residents (E_n) was calculated from E_c/A_0 as below:

$$E_n = 0.518 (E_c/A_0)^{1.204} \quad (6)$$

Solving for E_n and adding it to E_c gave a total summer effort $E_s = E_c + E_n$ of 4182 angler hrs or a unit area effort of 62.5 angler hrs ha^{-1} . As there is a winter fishery on the lake, winter effort (E_w) was calculated from

$$E_w = 0.811 (E_s/A_0)^{0.962} \quad (7)$$

and added to E_s to give an annual total effort (E_a) of 7080 angler hr or 106 angler hr $\text{ha}^{-1} \text{yr}^{-1}$.

Total harvest (H'') in $\text{kg ha}^{-1} \text{yr}^{-1}$ was calculated from the morphoedaphic index ($\text{MEI} = \text{TDS}/\bar{z}$, which was 1.45 g.m^{-4} for Harp Lake), and the calculated E_a :

$$\ln H'' = 0.295 \ln (\text{MEI}) - 0.006 (\ln \text{MEI})^2 + 0.654 \ln (E_a/A_0) - 1.717 \quad (8)$$

The calculated annual fish harvest (wet weight) was 282 kg yr^{-1} .

Harvey (personal communication) indicated that wet to dry ratios of fish from Harp Lake was 4:1. TP concentrations of whole fish average 4% of dry weight (Goodyear and Boyd 1972, Kitchell et al. 1975). Use of these conversion factors indicates that anglers annually remove an estimated 56 kg (dry weight) of fish from Harp Lake containing 2.1 kg of TP. As natural sources alone supply about 70 kg of TP to the lake each year (Table A13, in the Appendix) the influence of angling on TP budgets is small. Angling is probably of less importance in the other study lakes which have less development and poorer winter access.

Two mechanisms related to fish may re-introduce TP that is removed by anglers to the lake. About 30% of E_a is attributable to summer fishing by cottagers (E_c/E_a equals 2049/7080). Hence about 30% of the removed TP may be redeposited as domestic waste within the Harp Lake watershed. If this fraction is ultimately returned to the lake then the net TP lost from the lake through angling is reduced by 30% to 1.5 kg.

TP lost from the lake by non-resident angling is to some

extent balanced by fish stocking. From 1947 to 1981, Harp lake was stocked with speckled, brook and/or lake trout on thirteen occasions. Between 1976 and 1980, when natural TP inputs were measured during this study, 75.9 kg (wet weight) of lake trout were introduced into Harp Lake (Table 34). Using conversion factors identical to those above indicates that an average of 0.14 kg of TP per year was introduced into the lake in this way. Three other study lakes were stocked at somewhat lower intensities (Table 34). These calculations indicate that anglers irreversibly removed an order of magnitude more TP (1.5 kg) than was replaced by planting fish in Harp Lake.

ii) LOSS OF TP BY INSECT EMERGENCE

A second sink for TP in shield lakes is loss of TP in emerging insects. Davies (1980) developed a relationship between J_T/Q (the annual total supply of phosphorus to a lake divided by the outflow volume) and $\bar{B}$ (the lake average dry biomass of emergent Diptera, in $\text{g m}^{-2} \text{yr}^{-1}$). The measured annual natural total phosphorus supply to Harp Lake equals about 70 kg, and the average outflow volume equals about $2.8 \times 10^9 \text{ L yr}^{-1}$; hence, J_T/Q averages 25 mg m^{-3} for Harp Lake. Davies reported a relationship between $\bar{B}$ and J_T/Q :

$$\bar{B} = 0.182 + 0.00315 (J_T/Q) \quad (9)$$

From this, the average biomass of Diptera emerging from Harp Lake was estimated as 0.261 $\text{g m}^{-2} \text{yr}^{-1}$. This yields a total emergence of 174 kg of Diptera ($A_0 = 66.9 \times 10^4 \text{ m}^2$).

Vallentyne (1952) reported an average TP content of adult emergent insects of 0.24% of fresh weight, about 2.4% of dry weight assuming (Davies, pers. comm.) they are 90% water. The 174 kg of emerging Diptera represents a loss of 4.2 kg of P from Harp Lake, a significant quantity in comparison with other sinks such as outflow loss (≈ 15 kg) or fish harvest (2.0 kg).

This loss is only temporary, however. Davies (pers. comm.) stated that virtually all emerging adult female Diptera oviposited and subsequently died on the surface of the lake from which they emerged. Males usually also died near or over the water; hence, Dipteran emergence probably represents a very small net TP loss from Harp, or any of the study lakes.

Table 34. Contribution of TP to the "A" lakes from recent (post-1976) stocking of lake trout. Stock records obtained from Fisheries Branch, Ontario Ministry of Natural Resources.

Lake	Number of Fish Stocked	Years of Stocking	Total Weight of Stocked Fish (kg wet wt.)	TP Supplied in Stocked Fish ² (kg)
Blue Chalk	1500	1976-1978	17.7	0.13
Harp	6100	1976-1980	75.9 ¹	0.56
Jerry	6000	1977-1981	51.8	0.38
Red Chalk	5100	1976-1980	55.9	0.41

¹ assuming 2000, 14 month-old fish planted in 1979 had same average individual weight as 15 month-old fish planted in 1980.

² calculated assuming 4:1 ratio of wet:dry weight and P content of 3.7% dry weight (see text).

iii) NUTRIENT SUPPLY BY BIRD POPULATIONS

Large nesting colonies of birds may have a substantial impact on lake nutrient dynamics. In fact, the term "guano-trophic" has been coined to describe lakes or ponds whose eutrophic characteristics are attributable to nutrient inputs from birds (Leentvaar 1967, Paloumpis and Starret 1960, Brinkhurst and Walsh 1967). In Ontario, four bird species aggregate in large colonies – herring gulls, ring-billed gulls, Caspian terns and common terns. As none of the "A" lakes had gull or tern nesting colonies, it may safely be assumed that bird faeces was not a significant source of nutrients to these lakes. There was a substantial (>100 birds) nesting colony of ring-billed gulls on Gullfeather Lake, one of the "B" lakes, and clearly an appropriately named lake. The impact of an unmeasured source of nutrients to Gullfeather Lake, attributable to bird faeces, will be discussed in Section V.

iv) NUTRIENT SUPPLY BY FOREST LITTER

Airborne forest litter, especially leaf fall from deciduous forests, may introduce TP to lakes. Hanlon (1981) reported an average input of 260 g dry weight of litter m⁻¹ of wooded lake shoreline for four lakes in various parts of the world. Inputs ranged from a low of 135 g m⁻¹ for Welsh Llyn Frennoch to a high of 354 g m⁻¹ for Mirror Lake in New Hampshire. Assuming forest was continuous in the "A" lake watersheds, leaf-litter inputs were estimated using the mean input given by Hanlon. Estimates ranged from 1010 to 2020 kg of airborne vegetation litter per lake per year (Table 35). From an average TP concentration of leaves of 0.16% dry weight, calculated from Ovington (1956), leaf litter may constitute an estimated 1.6 to 3.2 kg of TP to the study lakes annually. The source is very small in comparison with total measured inputs, e.g., 3% in Harp Lake (Table 35). Leaf litter may become an important source of TP for lakes with very small watersheds, but it may be ignored for the "A" lakes.

Table 35. Estimated total input of airborne vegetation litter to the "A" study lakes. Estimate calculated from shoreline length (L) assuming input is 260 kg km⁻¹ of dry material (Hanlon 1981) and TP content of leaves is 0.16% of dry weight (Ovington 1956). Ratio of litter TP supply to natural TP supply is included.

Lake	L (km)	Total Airborne Litter Input		Ratio
		dry matter (kg yr ⁻¹)	TP (kg yr ⁻¹)	
Blue Chalk	4.6	1190	1.9	0.08
Chub	3.9	1010	1.6	0.04
Dickie	7.8	2020	3.2	0.04
Harp	4.6	1190	1.9	0.03
Jerry	4.6	1190	1.9	0.02
Red Chalk	4.8	1240	2.0	0.04

8. LOSS OF PHOSPHORUS FROM THE STUDY LAKES – SEDIMENTATION

Phosphorus is lost from the water column of lakes by two major processes, outflow and sedimentation. The loss of TP from the "A" lakes via outflow was measured accurately by quantification of the discharge (outflow volume per unit time section III.B.2) and the nutrient content of

the outflow. The flux of TP to the sediments of lakes is much more difficult to measure; unfortunately, the sediments are usually a more important sink for TP than is loss by outflow. Because of its importance, three independent measurements of TP accumulation in the study lakes' sediments were made.

i) RETENTION OF TP – ESTIMATES USING THE TP MASS BALANCES

The retention coefficient of a lake for TP (R_p) is defined as the fraction of the total input of TP to the lake that is not lost by outflow. If the concentration of TP in the lake is at steady-state (i.e. the concentration is not changing other than on a short-term or seasonal basis), then the retention coefficient is equivalent to the portion of the input lost to the lake's sediments:

$$R_p = \frac{J_T - J_o}{J_T} \quad (10)$$

where J_T = total input,
 J_o = loss by outflow.

For the undeveloped lakes (Jerry and Red Chalk Lakes), the total input of TP (other than that contributed by minor sources – Section III.B.7) was measured directly. The TP concentrations in the lakes were at steady-state (Section III.B.10). Therefore, the retention, or sedimentation rate of TP can be explicitly calculated for these lakes by dividing the mass of TP retained by the sediment area. These calculations are outlined in Table 36.

In Jerry Lake, between 42 and 73% (100 R_p) of the TP input (average of 60%) was lost to the sediments in the individual four years of study, while in Red Chalk Lake, the retention ranged from 47 to 79%, averaging 61%. The retention coefficients and derived sedimentation rates were highly variable from year-to-year, indicating that long-term measurements of mass budgets are essential for accurate calculation of average sedimentation rates.

The TP retention coefficients and sedimentation rates could not be measured directly for the developed lakes because their anthropogenic, and therefore their total input of TP was not known. However, empirical models

Table 36. Sedimentation of TP in undeveloped lakes calculated from the mass balances. Total input (J_T), loss by outflow (J_o), amount retained, sedimentation rate (S), and retention coefficient (R_p) are reported.

Lake	J_T	J_o	Retained	S	R_p
		(kg yr ⁻¹)		(mg m ⁻² yr ⁻¹)	
Jerry					
1976-77	112.3	29.9	82.4	164	0.734
1977-78	79.5	27.7	51.8	103	0.652
1978-79	102.3	45.9	56.4	113	0.551
1979-80	81.7	47.4	34.3	68	0.420
Average	94.0	37.7	56.3	112	0.598
Red Chalk					
1976-77	57.8	12.2	45.6	80	0.789
1977-78	56.2	20.8	35.4	62	0.630
1978-79	50.8	24.6	26.2	46	0.516
1979-80	49.6	26.3	23.3	41	0.470
Average	53.6	21.0	32.6	57	0.608

relating R_p to independent variables (often hydrologic parameters) have been developed and validated for other Precambrian lakes (Kirchner and Dillon 1975, Dillon and Kirchner 1975b, Nurnberg 1983). The information collected for the two undeveloped lakes was therefore used to validate existing models which may then, in turn, be used to predict R_p and S for the developed lakes. This approach is discussed in Section III.B.10.

ii) ESTIMATES USING SEDIMENT TRAPS

In theory, it should be possible to measure the sedimentation rate of TP (or any other substance) by collection of sedimenting particles in containers (sediment traps) positioned above the sediment-water interface. Sediment traps were used to estimate the whole-lake sedimentation of TP in Blue Chalk Lake. Traps were located at eleven sites on the lake over a period of four hydrologic years. Sedimentation rate was calculated at each station for each year. The average of all stations was used as a whole-lake sedimentation rate estimate (T).

Rates of sedimentation of TP are given in Table 37. The TP sedimentation rates calculated in this way ranged from 442 to 640 $\text{mg m}^{-2} \text{yr}^{-1}$, with a mean value of 504 $\text{mg m}^{-2} \text{yr}^{-1}$. These values of T are much greater than those rates (S) calculated from the TP mass balance of the lake. Since the anthropogenic TP inputs were unknown, a range in S was calculated each year assuming that 0 and 100% of the estimated potential anthropogenic input of TP reached the lake. Since this potential input was relatively small (between 17 and 31% of the "natural", measured input), the range in S was not great, averaging 39 to 51 $\text{mg m}^{-2} \text{yr}^{-1}$. Rates of sedimentation of TP, measured using sediment traps, were therefore 12 to 17 times greater than that calculated from the measured mass balances. Since T was an order of magnitude greater than the total possible input to the lake (assuming 100% of the estimated potential anthropogenic TP enters the lake), sediment traps clearly do not measure the sedimentation of TP properly.

The usefulness of the sediment trap methodology has been intensely debated (e.g. Hargrave and Burns 1979,

Table 37. Comparison of TP accumulation ($\text{mg m}^{-2} \text{yr}^{-1}$) in the sediments of Blue Chalk Lake measured using sediment traps (T), and calculated from the mass balance (S). Standard errors in brackets.

Year	Sedimentation		T/S
	T	S ²	
1976-77	— ¹	61.4 – 72.8	—
1977-78	640	40.4 – 51.8	12 – 16
1978-79	475	31.0 – 42.5	11 – 15
1979-80	457	23.3 – 34.7	13 – 20
1980-81	442	— ¹	—
Average	504 (46)	39.0 – 50.5 (8.2) (8.2)	12 – 17

¹ no data

² low end of range calculated assuming input from anthropogenic sources to the lake was 0; the high end of the range calculated assuming all of the potential input from anthropogenic sources reached the lake.

Gardner 1980a, 1980b). Similar findings to this study have been reported by Fuhs (1975), who measured a 10-fold difference between TP accumulation in traps and accumulation calculated from the mass balance for Canadarago Lake, New York.

Several factors may contribute to the overestimation of sedimentation of TP by sediment traps. It has been suggested that collection of particulate material in sediment traps interrupts the normal mineralization-production cycling of elements. Other possible phenomena which would result in over-collection of material include the "snow-fence" effect, whereby the horizontal velocity of particles passing the trap-mouth is reduced, resulting in a greater catch. Resuspension of particles already in the surficial sediments is also a common occurrence, especially at spring and fall overturn, and may lead to over-collection of particles in the traps.

The results of this study suggest that sediment traps do not give a quantitative measure of sedimentation rate of TP. Sedimentation rates calculated from sediment traps are therefore not used in the mass balance models.

iii) ESTIMATES USING SEDIMENT CHRONOLOGIES

The sedimentation rate of TP in the lakes was estimated in a third way by using the historical (or paleolimnological) record contained in lake sediments. Recent development of relatively simple sediment dating techniques allows determination of the age of sediment layers deposited in the last few hundred years. Thus, the accumulation rate of matter in the lake's sediments may be calculated. If the nutrient content of these sediments is known, the accumulation rate of the nutrient can therefore be calculated.

To establish the chronologies and thus, the sediment accumulation rates, of lake sediments over the last approximately 100 years, Pb-210 has been shown to be very useful. The rationale for its use has been reviewed by Koide et al. (1973), Robbins (1978) and Krishnaswami and Lal (1978).

Lead-210 is an intermediate product of the U-238 decay series. It is introduced into lakes' sediments by two processes. The decay of Ra-226 in the earth's crust results in the exhalation of Rn-222 to the atmosphere. This isotope rapidly decays to Pb-210, which is, in turn, rapidly removed by wet precipitation and dry fallout, the mean residence time in the atmosphere being only 5-10 days. The residence time of Pb-210 in lakewater is also short because of its affinity for particulates. For the same reason, input of Pb-210 to a lake from its watershed is usually negligible unless transport of particulates (erosion) is an important process. This is very unlikely to be important in Precambrian areas.

The second process resulting in the accumulation of Pb-210 in lake sediments is the decay of the parent Ra-226 in the lakewater and sediments themselves. Since the Pb-210 produced in this way is in secular equilibrium with the Ra-226 in the sediments and water column, this additional activity, which is termed the background or supported Pb-210 activity, will be constant.

Since the flux of Pb-210 to the lake's sediments from the atmosphere is constant and the Pb-210 is not mobile in the sediments, then the change of Pb-210 concentration in the sediments is a function of only radioactive decay and/or change in accumulation rate of lake sediments. Therefore, if no change in sedimentation rate has occurred and there are no processes occurring that result in mixing of the surficial sediments, then the activity of Pb-210 will be log-linear with depth (i.e. with time) until the supported Pb-210 level is reached. Calculation of the sediment accumulation rate (SAR) from the Pb-210 profile is done using the relationship:

$$A(w) = A(m)e^{-\tau w} + A(s) \quad (11)$$

where: $A(w)$ is the Pb-210 activity at any depth defined by w where the mixing zone w depth of supported Pb-210 level

w is the depth scale measured as cumulative dry weight per unit area

$A(m)$ is the activity of Pb-210 at the bottom of the mixing zone (or at the sediment surface if there is no mixing zone)

$A(s)$ is the activity of supported Pb-210

τ is the Pb-210 decay constant ($\tau = \ln 2/t_{1/2} = 0.0311 \text{ yr}^{-1}$)

The slope, k , of the regression of $\ln[A(w)-A(s)]$ on w is related to the accumulation rate of sediments (SAR in dry weight area⁻¹ time⁻¹) by the decay constant:

$$k = \tau/\text{SAR} \quad (12)$$

where $\text{SAR} = w/t$

Therefore, $\text{SAR} = \tau/k \quad (13)$

It must be emphasized that sediment accumulation rates measurements are site-specific; the rate measured at one location cannot be extrapolated to the entire lake or to another site within the lake. However, a recent methodology for establishing whole-lake sediment and nutrient accumulation rates has been proposed by Evans (1980), further developed by Dillon and Evans (1982), and utilized by Evans and Dillon (1982) and Evans et al. (1983).

Table 38. Anthropogenic whole-lake Pb burdens for ten lakes in the study area, including all of the "A" lakes. Standard error of the mean in brackets. "n" is the number of cores from which the whole-lake Pb burden was calculated.

Lake	Pb burden (mg m ⁻²)	n
Blue Chalk	541	67
Chub	645	43
Clear ¹	624	48
Crosson ¹	627	34
Dickie	608	39
Harp	814	107
Heney ¹	732	48
Jerry	733	44
Plastic ¹	768	36
Red Chalk	676	92
Mean	677 (26)	

¹ additional lakes studied as part of the Acid Precipitation in Ontario Study.

Dillon and Evans (1982) demonstrated that atmospheric deposition was the only significant source of stable Pb to Precambrian lakes in southern Ontario, and that the Pb was almost entirely of anthropogenic origin (combustion of fossil fuels). Lead deposited on the terrestrial portions of the watersheds was not mobile and did not contribute to that found in the lake. Furthermore, the lakes were extremely efficient sinks or traps for the Pb deposited on their surface; all Pb input to the lakes was retained in the lakes' sediments. The result of these facts is that all the lakes' sediments contain the same mass or "burden" of Pb (per unit area). Whole-lake Pb burdens for lakes in the study area including all of the "A" lakes are shown in Table 38. The mean burden (680 mg Pb m⁻²) is an estimate of the average amount deposited over the entire study area. Since precipitation from the atmosphere occurs uniformly over the lakes' surfaces over the long-term, there should be no spatial variability in total Pb burden in the lakes' sediments if there was uniform sedimentation of particulate matter. However, sedimentation is not uniform, because of redistribution of sediments to the deeper parts of each lake. The anthropogenic Pb must also be redistributed in a similar, perhaps identical, pattern. Therefore, a comparison of the sediment Pb burden at a particular site to the whole-lake Pb burden must reflect the differential movement of sedimenting particles, and hence the distribution of sediments.

This ratio thus may be employed as a correction factor to estimate whole-lake accumulation from a site-specific measurement of the sediment accumulation rate. Combined with the TP concentration of the sediments at that site the whole-lake TP accumulation rate may then be calculated.

A detailed calculation is shown in Table 39. The sediment accumulation rate measured at four hypolimnetic sites in Jerry Lake was corrected using the whole lake site-specific Pb burden ratio. The resulting average SAR (69 g m⁻² yr⁻¹) can be combined with the TP concentration of the sediments at the lake's mean depth (1.43 mg g⁻¹, from the relation $[TP] = 0.080 (\text{depth}) + 0.435$) to give an estimated TP accumulation for Jerry Lake of

Table 39. Sediment accumulation rate (SAR(s)) measured at 5 sites in Jerry Lake using the Pb-210 method, Pb burden at the same sites, ratio (F) of whole-lake Pb burden to site-specific Pb burden, corrected sediment accumulation rate (SAR(w)), and TP concentration in the upper 3 cm of sediment at each site.

Site Number	Station depth (m)	SAR(s) (g m ⁻² yr ⁻¹)	Pb burden (g m ⁻²)	F ²	SAR(w) (g m ⁻² yr ⁻¹)	[TP] (mg g ⁻¹)
1 ¹	4.8	465	637	1.151	535	0.99
2	11.0	91	965	0.760	64	1.33
3	17.3	118	1262	0.581	89	1.77
4	20.2	109	2139	0.343	63	1.67
5	32.3	173	1034	0.709	59	3.23
Average					69 (14) ³	

¹ littoral zone core - coarse sediments with very low organic content, excluded from subsequent calculations.

² whole-lake Pb burden of 733 mg m⁻² (Table 38).

³ standard deviation.

99 mg m⁻² yr⁻¹. Because the Pb-210 methodology gives an average sediment accumulation rate over a two or three-decade period, this calculation of TP accumulation is, in fact, an estimate of the long-term average.

This result compares favourably with the average value measured for Jerry Lake using the mass balance data (112 mg m⁻² yr⁻¹; Table 36).

The TP accumulation rates of the "A" lakes calculated using this methodology are reported in Table 40. The results calculated in this manner can only be accurately compared to the results calculated from the mass balances for the undeveloped lakes (Red Chalk and Jerry Lakes). The comparison is less favourable for Red Chalk Lake; 85 mg m⁻² yr⁻¹ vs. 57 mg m⁻² yr⁻¹.

In summary, independent estimates of TP accumulation in lakes' sediments can be made for the undeveloped lakes using the TP mass balances and for all lakes using the whole-lake sediment accumulation rates and sediment TP concentrations. Sediment traps do not provide meaningful measures of TP flux to lakes' sediments.

Table 40. TP accumulation rates for the "A" lakes calculated from the Pb burden-corrected sediment accumulation rates and the TP concentration of the sediments.

Lake	TP accumulation rate (mg m ⁻² yr ⁻¹)
Blue Chalk	37
Chub	81
Dickie	74
Harp	85
Jerry	99 ¹
Red Chalk	85 ²

¹ compared to 112 mg m⁻² yr⁻¹ (Table 36) calculated from the mass balance.

² compared to 57 mg m⁻² yr⁻¹ (Table 36) calculated from the mass balance.

9. TOTAL PHOSPHORUS MASS BALANCES

As discussed in previous sections, the major components of the TP balances of a lake are the flux from the terrestrial watershed either in streamflow or as ephemeral runoff draining the "ungauged" areas of the watershed, the flux from the atmosphere to the lakes, the loss of TP from the lakes by outflow or by sedimentation to the lake bottom, and possibly the input of TP derived from anthropogenic sources (mainly septic systems) into the developed lakes. In this section, the TP mass balances of the "A" lakes are discussed in order of decreasing potential contribution of anthropogenic TP sources. The mass balances are summarized in Tables A13-A18 and presented in Figures 5 to 10.

i) HARP LAKE

Over the four-year survey period, the six gauged inflows to Harp Lake contributed an average of 55% of the measured (i.e., the natural) TP input to the lake, with individual streams providing from 1% (inflow 3A) to 32% (inflow 5) of the total. The amount of TP contributed by

any stream varied substantially from year to year, with differences of greater than 100% measured in one case (inflow 6A). The relative importance of the different streams was very consistent from year to year, in part because the larger watersheds provided more runoff and therefore more TP. However, small watersheds may be important contributors to the TP content of lakes. For example, two inflows contributed more TP to Harp Lake than inflow 3, but none had as high a volume-weighted TP concentration (Table 41). Therefore, inflow 3 provided the most TP per volume of water displaced in Harp Lake of all inflows. On the other hand, inflow 3A provided the least amount of TP to the lake, and the least per unit volume of water displaced.

The ungauged area of the Harp Lake watershed contributed an estimated 8.5% (range 6.7-10.7%) of the total input. This amount is relatively small; that is, >90% of the "measurable" inputs were measured, providing confidence in the accuracy of the measurement of the TP inputs.

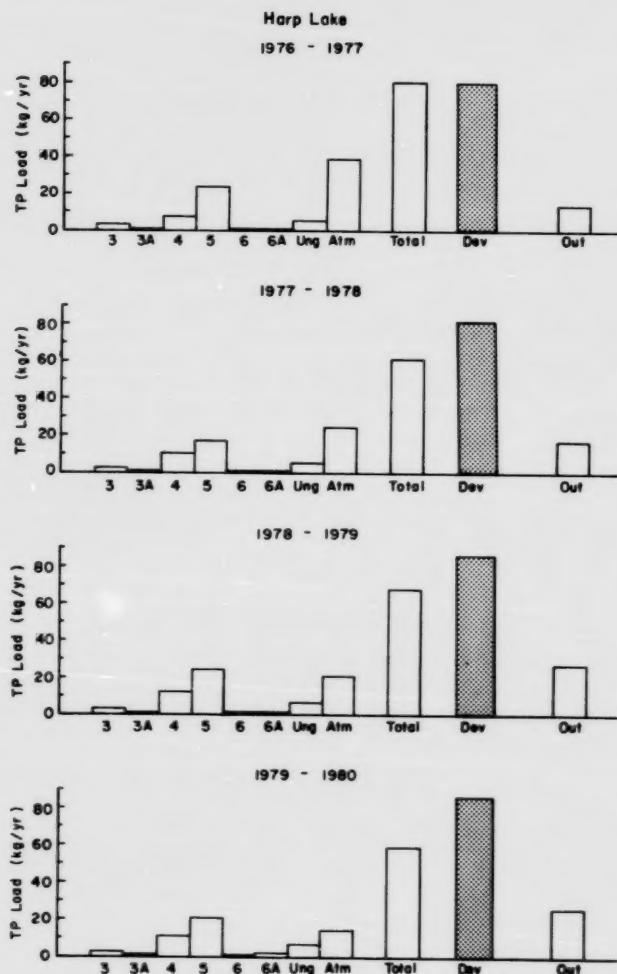


Figure 5 Total phosphorus loading to Harp Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

Atmospheric deposition was a very significant source of TP, providing $\approx 36\%$ of the measured input (range 25-48%) of TP during the study period. The volume-weighted concentration was much greater than that of any of the inflows (Table 41); therefore, atmospheric deposition provided the greatest net input of TP (amount of TP supplied per unit of lake volume displaced) of all natural sources.

The potential supply of TP from anthropogenic sources (section III.B.6) was greater than the sum of all natural inputs (326 kg TP in four years vs. 270 kg from natural sources). If all of the potential anthropogenic TP input actually reached Harp Lake, it would represent 55% of the total input, by far the most of any single source.

The loss of TP by outflow from Harp Lake was relatively small compared to even the natural inputs. Because the TP concentration in the lake was at steady-state during the study period, these results provide further evidence that loss by sedimentation is the major removal mechanism for TP from the lake.

Table 41. Input of TP to Harp Lake per unit volume of water from the natural sources (J/Q) and total input from each source (J). Results are mean values for 1976-80 period.

Source	J/Q J (mg m ⁻³)	J (kg yr ⁻¹)
Inflow 3	23.5	3.0
Inflow 3A	6.3	0.7
Inflow 4	15.0	10.2
Inflow 5	20.2	21.5
Inflow 6	13.7	0.9
Inflow 6A	14.4	1.0
Ungauged	15.9	5.8
Atmosphere	37.7	24.6

ii) DICKIE LAKE

The TP mass balance of Dickie was similar to that of Harp Lake in several respects. The five gauged tributaries contributed an average of 40% of the natural input of TP, while atmospheric deposition contributed 38%. The ungauged area provided a greater proportion of the TP input than in the case of Harp lake (22% vs. 8.5%).

The relative importance of the different tributaries was again different if the volume-weighted concentration was considered instead of the absolute input (Table 42).

Table 42. Input of TP to Dickie Lake per unit volume of water from the natural sources (J/Q) and total input from each source (J). Results are mean values for 1976-80 period.

Source	J/Q J (mg m ⁻³)	J (kg yr ⁻¹)
Inflow 5	28.3	5.3
Inflow 6	56.3	7.2
Inflow 8	14.1	3.7
Inflow 10	27.2	11.3
Inflow 11	17.6	8.1
Ungauged	30.0	7.5
Atmosphere	35.1	8.8

Inflow 6 contributed most to the TP balance according to the former methodology followed by inflows 5 and 10, while inflow 10 followed by inflow 11 contributed most TP in absolute terms.

The potential input of TP from anthropogenic sources was 87% of the natural TP input; that is, if the anthropogenic sources contributed their full potential input, they would constitute 46% of the total input.

Outflow loss of TP was small compared to natural input, again demonstrating the importance of sedimentation.

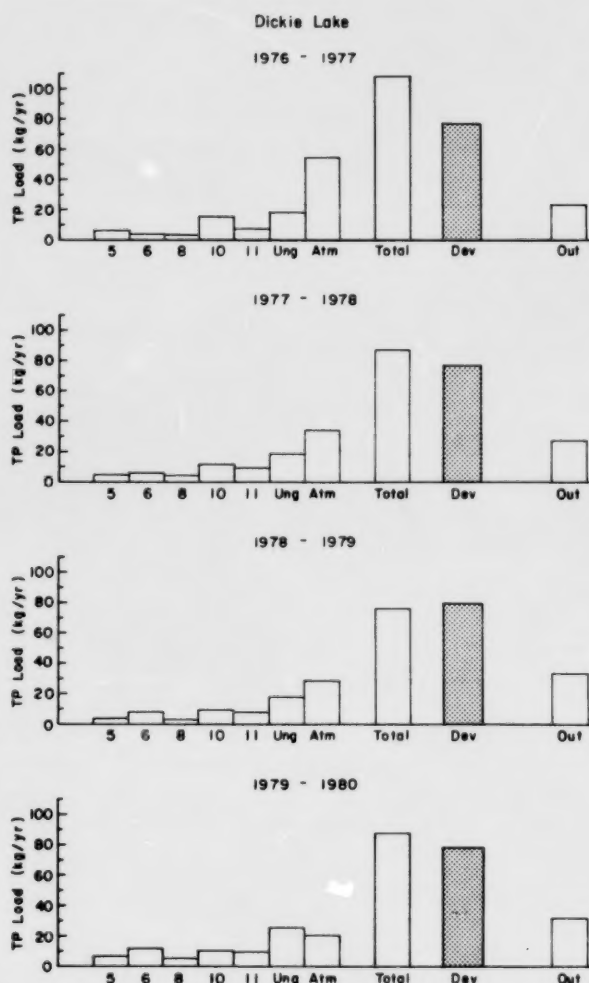


Figure 6 Total phosphorus loading to Dickie Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

iii) CHUB AND BLUE CHALK LAKES

The two gauged inflows of Chub lake contributed 50% of the natural TP input over the four-year period; atmospheric deposition contributed an additional 32%. The unmeasured portion of the TP input (18%) was comparable in importance to that of Dickie Lake.

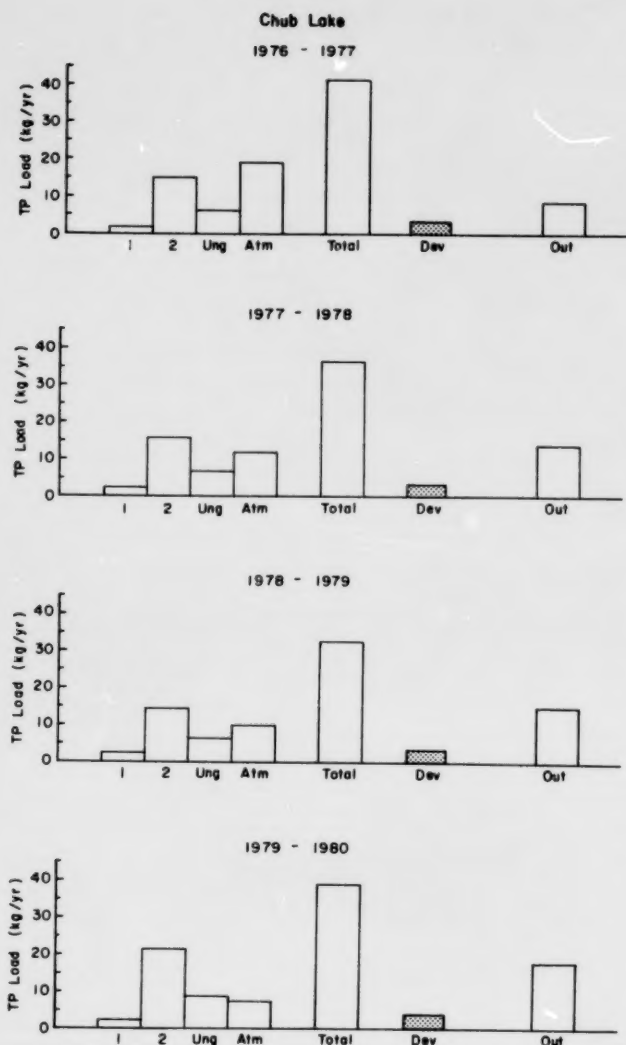


Figure 7 Total phosphorus loading to Chub Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

The potential input of TP from anthropogenic sources was much less important than in the cases of Harp and Dickie Lakes, averaging 7.3% of the measured natural input, or 6.8% of the total potential input of TP.

Loss of TP by outflow was about 37% of the natural input, with sedimentation again necessarily the more significant loss mechanism.

The one gauged tributary of Blue Chalk Lake provided only 2.2% of the natural input of TP. Atmospheric deposition provided by far the major portion of the TP input ($\approx 75\%$), while the ungauged area contributed an estimated 23%. Anthropogenic sources could potentially provide 23% as much TP as the combined natural sources (19% of the potential total input). About 20% of the natural input was lost by outflow.

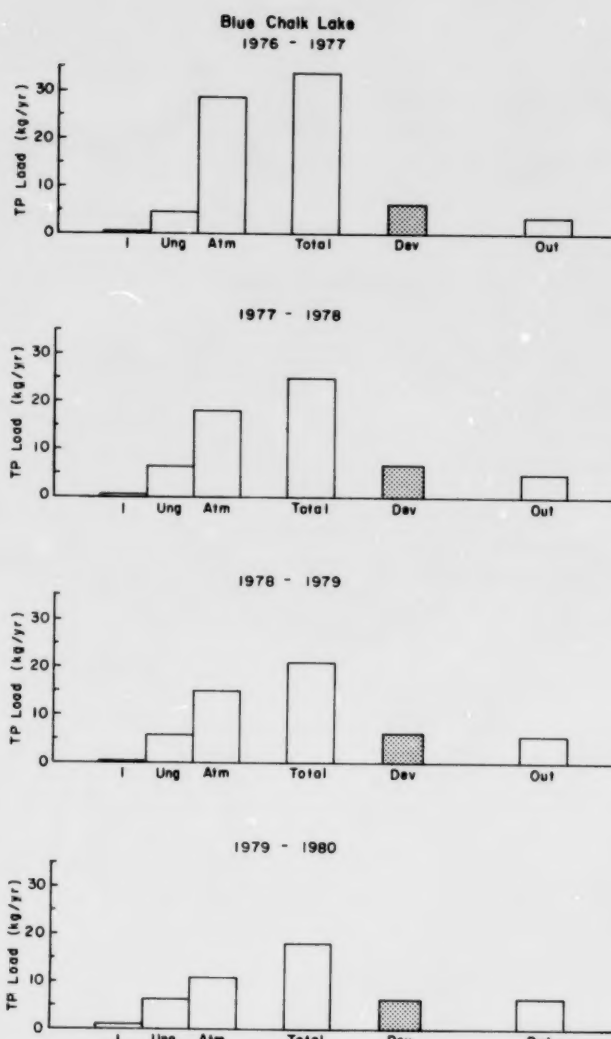


Figure 8 Total phosphorus loading to Blue Chalk Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

iv) RED CHALK AND JERRY LAKES

The four gauged tributaries of Red Chalk Lake contributed 42% of the natural input of TP; the outflow of Blue Chalk Lake contributed an additional 10%. Atmospheric deposition supplied 39% of the TP input, with the ungauged area supplying 9%. The potential contribution from the lakeshore development was insignificant.

The TP mass balance of Jerry Lake was similar to that of Red Chalk Lake; gauged tributaries were the most important source of TP (71%) followed by atmospheric deposition (20%), then by runoff from the ungauged portion of the watershed (10%). There was no potential input from anthropogenic sources.

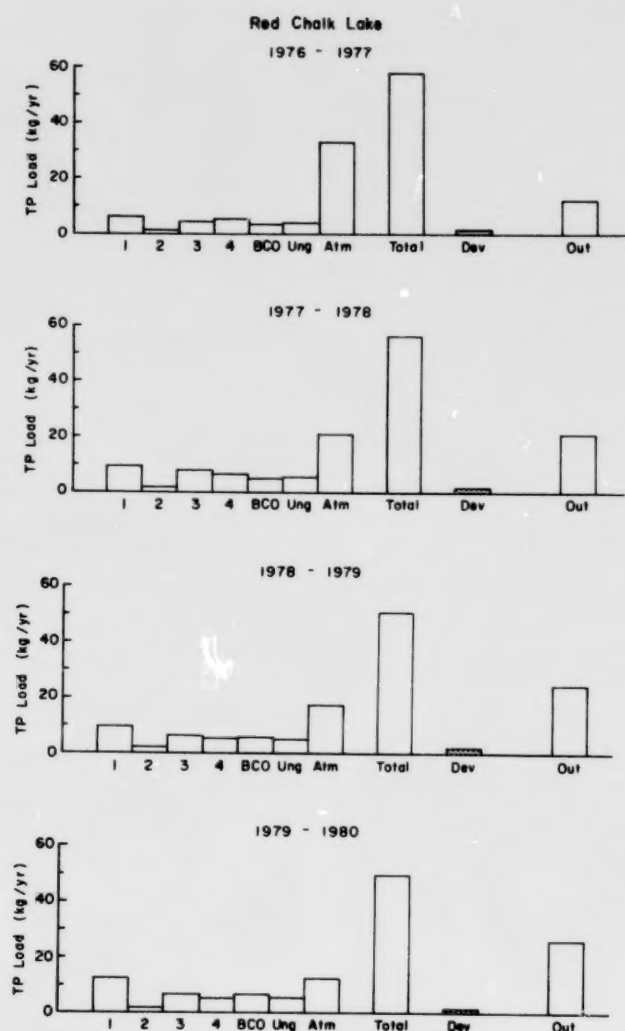


Figure 9 Total phosphorus loading to Red Chalk Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

v) COMPARISON OF THE TOTAL PHOSPHORUS MASS BALANCES

Based on consideration of the measured TP mass balances of the "A" lakes, several generalizations can be made.

The inflowing streams to the study lakes were of highly variable importance, providing as little as 0.4% of the natural input of TP in the case of Jerry Lake inflow 1 or 1% in the case of Harp Lake inflow 3A to as much as 69% (Jerry Lake inflow 3) and 44% (Chub Lake inflow 2).

The input of TP by atmospheric deposition was an extremely important flux; it was the single most important source for four of the six lakes in all years, and the most important in a fifth lake in one of the four years. Atmospheric contributions are most significant for lakes with small terrestrial watersheds.

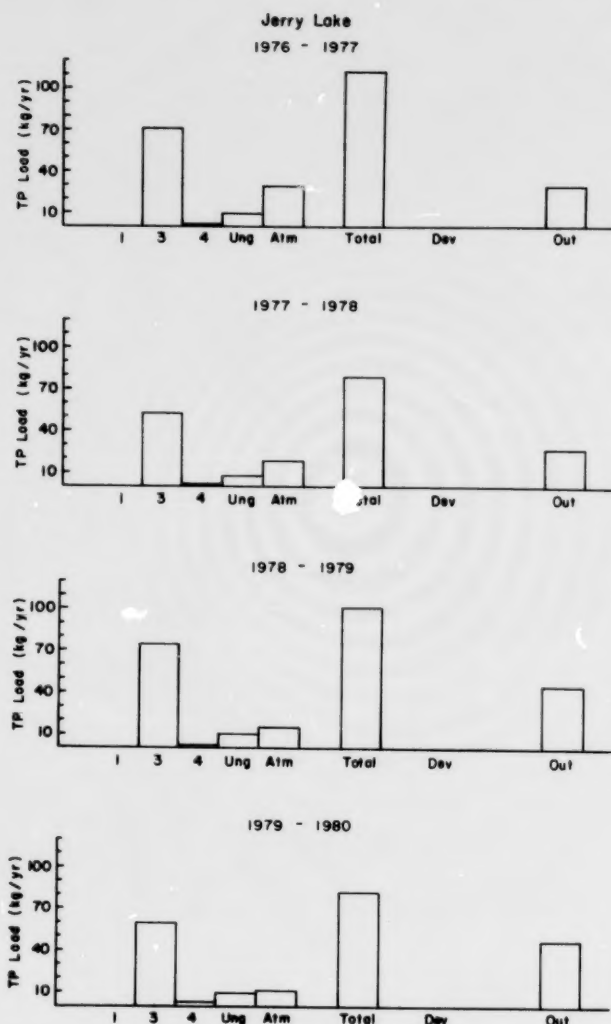


Figure 10 Total phosphorus loading to Jerry Lake, in kg/year for four hydrologic years (1976-1980). Inputs are measured streamflow (stream identification number), ungauged area (Ung), atmospheric loading to the lake surface (Atm) and calculated supply from waste disposal systems (Dev). The sum of the natural inputs (Total) and loss by outflow (Out) are also given.

Contributions of TP from the ungauged watershed areas were not very great, averaging 15% of the total natural input for the six "A" lakes. Therefore, the effect of uncertainty in the measurement of the TP input from these ungauged areas on the TP mass balances is not large.

Loss of TP from the lakes by outflow was always less important than loss by sedimentation, even if the potential anthropogenic inputs were not considered.

Year-to-year variability in individual components of the mass balances was great. The input of TP to each lake by atmospheric deposition varied by almost a factor of three over the four-year study period. Inputs from specific inflows varied by up to a factor of two, as did loss by outflow. The annual inputs of TP to the study lakes from

Table 43. Annual natural total phosphorus load (L_p) to each of the "A" lakes, 4-year average annual load, and coefficient of variation (c.v.) of annual loads for each lake. Potential inputs from lakeshore development are not included.

Lake	L_p (mg m ⁻² yr ⁻¹)					c.v. (%)
	1976-77	1977-78	1978-79	1979-80	1976-80	
Blue Chalk	68.4	50.2	42.3	36.4	49.3	28
Chub	128.0	112.7	100.9	120.5	115.5	10
Dickie	116.3	93.1	81.2	93.9	96.1	15
Harp	121.1	92.4	101.6	88.8	101.0	14
Jerry	224.2	158.7	204.2	163.1	187.5	17
Red Chalk	101.6	98.8	89.3	87.2	94.2	7
					mean	15

natural sources on an areal basis are summarized in Table 43. The mean annual loads ranged from 49 mg TP m⁻² yr⁻¹ (Blue Chalk Lake) to 188 mg TP m⁻² yr⁻¹ (Jerry Lake). Variability (expressed as the coefficient of variation) in the TP loads ranged from 7% (Red Chalk Lake) to 28% (Blue Chalk Lake), with a mean of 15%. Thus, the 95% confidence interval for the load was equal to the mean load $\pm$ 24% (range $\pm$ 11% to $\pm$ 45%). Results of one-way ANOVA (using z-scored data) indicated that there were highly significant differences in load between years ($p < 0.00002$). The measurement of TP mass balances for several years should be considered essential for development and validation of trophic status models.

In order to establish the importance of the hydrologic balance to the TP loads, the TP loads (L_p) were divided by the areal water load (q_s).

The resulting parameter (L_p/q_s) with units of mg m⁻³ is equivalent to the volume-weighted mean TP concentration of all the measured natural inputs (Table 44). Statistical analysis (one-way ANOVA) indicated that there were still significant differences in L_p/q_s ($p < 0.01$) between years. However, the atmospheric deposition of TP (Section III B-4) was much greater in the first 12-month period (1976-77) than in the subsequent three annual periods. When the values of L_p/q_s for the last three years were compared, there were no significant differences between years ($p > 0.05$). In summary, the volume-weighted mean concentration of TP in the inputs varied only in the one year that atmospheric deposition of TP was very high.

Table 44. Annual total phosphorus load divided by areal water load (L_p/q_s) for each of the "A" lakes. Mean and coefficient of variation are reported. Potential TP inputs from lakeshore development are not included in the calculation.

Lake	L_p/q_s (mg m ⁻³)					c.v. (%)
	1976-77	1977-78	1978-79	1979-80	1976-80	
Blue Chalk	52.6	34.9	21.7	17.4	31.7	16
Chub	47.8	26.7	19.8	23.7	29.5	13
Dickie	57.0	39.5	28.1	30.1	38.7	13
Harp	45.7	24.4	19.1	17.5	26.7	13
Jerry	31.3	20.7	18.5	18.3	22.2	6
Red Chalk	23.2	19.1	12.6	12.5	16.8	5
					mean	11

Table 45. Total input of TP to the "A" lakes from natural sources (J_{NAT}), potential input from anthropogenic sources (J_A), ratio of J_A to total potential input calculated from measured mass balances, and original estimate (E) of this ratio from Table 1.

Lake	J_{NAT}	J_A	$J_A/(J_A + J_{NAT})$	E
	kg yr ⁻¹			
Blue Chalk	24.4	5.7	0.19	0.27
Chub	37.2	2.7	0.07	0.26
Dickie	89.6	77.7	0.46	0.54
Harp	67.6	81.5	0.55	0.61
Jerry	94.0	0.0	0.0	0.0
Red Chalk	53.6	0.6	0.011	0.04

The potential contribution of the anthropogenic sources to the TP mass balances are outlined in Table 45. The results are, except in the case of Chub Lake, very similar to the original estimates (Table 1) made at the start of the study on the basis of existing models. Shoreline development may have a major effect on the trophic status of Harp and Dickie Lakes, an intermediate effect on Blue Chalk Lake, a minor effect on Chub Lake and no effect on Red Chalk or Jerry Lake.

10. THE EFFECTS OF SHORELINE DEVELOPMENT ON THE PHOSPHORUS CONCENTRATIONS OF THE STUDY LAKES

As outlined in Section III. A, the purpose of this section is to evaluate the effects of shoreline development on the trophic status of the "A" lakes. Since the trophic status of a lake is governed by its total phosphorus concentration, this evaluation is based on the contribution of shoreline development to the TP concentrations of the lakes.

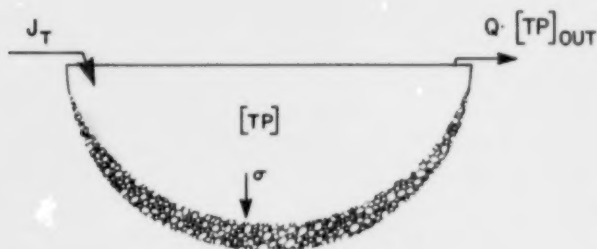
As shown in Figure 1, the TP concentration of a lake is a function of the lake's hydrologic budget, TP mass balance and morphometry. Over the past 10-15 years, numerous models (e.g. Vollenweider 1969, Dillon and Rigler 1974, Vollenweider 1975, Reckhow 1977, Walker 1977, and others) have been employed to interrelate these parameters, usually with the explicit purpose of predicting the effects of changes in the input of TP to the lake on its trophic status. In the case of the "A" lakes, the entire input of TP is known for the two undeveloped lakes (Jerry and Red Chalk), but is not known for those with shoreline development. If the concentration of TP in the study lakes is at steady-state (i.e. is not changing – a condition tested in a later section), then a mass balance model can be validated or calibrated using the undeveloped lakes. The validated model may then be applied to the developed lakes to derive a measure of the entire TP input. The difference between the TP input derived in this manner and that part of it that was measured (natural TP input) is attributable to the anthropogenic input of TP, which in turn may be related to the extent of shoreline development.

i) TP CONCENTRATION MODELS

The earliest mass balance models that were used to make predictions about the chemical content of lakes were derived by Vollenweider (1969) and others (reviewed in

Dillon 1974). These models treated lakes as completely mixed reactors, in which chemical inputs were mixed instantaneously throughout the entire water column. Substances were assumed to be supplied at a constant rate from atmospheric deposition, runoff and point sources, and lost at a constant rate via the lake outlet and by sedimentation (Figure 11a). In the simplest case, loss by sedimentation did not occur (i.e. the substance was conservative as outlined in Section III.B.3). However, as discussed in Section III.B.8 and III.B.9, sedimentation from the water column is the major mechanism for loss of TP in the study lakes and must be considered in model formulation.

(a)



(b)

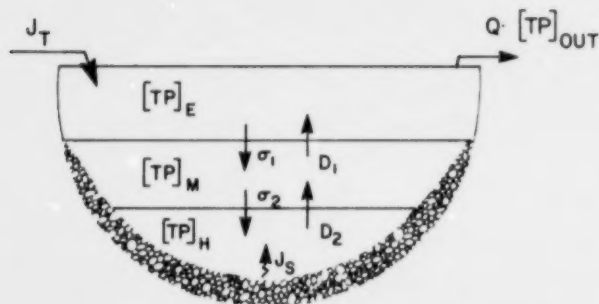


Figure 11 Schematic of TP flux in a lake. In the unstratified model (a), TP is lost by outflow ($Q[TP]_{OUT}$) and by sedimentation (σ). In the "three-box" model (b), TP is transported between layers by diffusion (D_1 , D_2) and sedimentation (σ_1 , σ_2). Release of TP from the sediments (J_s) may contribute TP to the hypolimnion.

Therefore in the case of TP, this simple model can be expressed in mathematical format as:

$$\frac{d[TP]}{dt} = \frac{J_T}{V} - \frac{Q[TP]_{OUT}}{V} - \sigma [TP] \quad (14)$$

where:

$[TP]$ = concentration of TP in the lake at time t

J_T = total input of TP to the lake from external sources per unit time

Q = volume of water flowing out of the lake per unit time

V = lake volume

$[TP]_{OUT}$ = concentration of TP in the lake's outflow

σ = sedimentation rate constant (time^{-1})

If $[TP]_{OUT} = [TP]$, then the concentration of TP at any time, t , is given by:

$$[TP] = J_T (1 - e^{-\phi t}) / V\phi + [TP]_0 e^{-\phi t} \quad (15)$$

where: $[TP]_0$ = the concentration of TP at $t = 0$
 ϕ = the sum of the loss rate constants
 $= Q/V + \sigma$

At steady-state (i.e. as $t \rightarrow \infty$):

$$[TP]^I = J_T / V\phi \quad (16)$$

where: $[TP]^I$ = concentration of TP in the lake at time $t \rightarrow \infty$

Alternatively, loss by sedimentation can be expressed in terms of a sedimentation velocity, v , describing the downward flux of TP:

$$\frac{d[TP]}{dt} = \frac{J_T}{V} - \frac{Q[TP]_{OUT}}{V} - \frac{v \cdot A_s [TP]}{V} \quad (17)$$

where: A_s = the area of the plane through which sedimentation occurs

$A_s = A_0$, the lake area, in the simplest case

$$\text{Thus, } [TP]^I = J_T / (Q + v \cdot A_0) \quad (18)$$

A third approach to describe the sedimentation of TP is use of the retention coefficient (Dillon and Kirchner 1975b). It can be shown that at steady-state:

$$[TP]^I = J_T (1 - R_p) / Q \quad (19)$$

$$\text{or } [TP]^I = L (1 - R_p) / q_s \quad (20)$$

where: R_p = the fraction of the total input of TP not lost via the outflow, and

q_s = areal water load = Q/A_0

L = areal loading rate of TP = J_T/A_0

It can also be shown that equations 17 and 19 are related in the following manner:

$$R_p = v / (v + Q/A_0) \quad (21)$$

$$\text{or } R_p = v / (v + q_s) \quad (22)$$

Empirical calibration of equation 22 with data collected for oligotrophic, Precambrian lakes (Dillon and Kirchner 1975) indicated that the settling velocity, v , averaged 12.4 m yr^{-1} (corrected from 13.2 m yr^{-1} in the original paper). Therefore, the simplest version of the TP mass balance model can be written as:

$$[TP]^I = J_T / (Q + 12.4 A_0) \quad (23)$$

$$\text{or } [TP]^I = L / (q + 12.4) \quad (24)$$

In this simple model formulation, the assumption is made that the concentration of TP in the lake is at steady-state. This assumption was tested for the study lakes for the 1976-80 period using TP data (Table 46) collected during five time periods (spring overturn, summer stratified (whole lake) period, fall overturn, ice-free period and annual). For each time period, results of two-way ANOVA showed that there were no significant differences ($p < 0.05$) in TP concentration between years. Therefore, the steady-state version of the model is applicable and the four years can be considered as replicate measurements. This result can be explained on the basis of the response time of the lakes to a change in the input of TP. Since the water replenishment times of the lakes are greater than one year (Table 14), changes in input are not immediately manifested. Measured TP concentration in the study lakes at spring overturn in the subsequent three years (1980, 1981, 1982) were similar to those measured in the 1976-1980 period (Table 47), confirming that the assumption that the TP concentrations were in steady-state is valid.

Table 46. Total phosphorus concentration (mg m^{-3}) during spring overturn (subscript SO), summer stratification (subscript SS), fall overturn (subscript FO), ice-free (subscript IF), and annual (subscript AN) periods for "A" lakes for each of 4 12-month periods and for 1976-80. Concentrations in the stratified, ice-free and annual periods are whole-lake, volume-weighted averages.

Lake/year	[TP] _{SO}	[TP] _{SS}	[TP] _{FO}	[TP] _{IF}	[TP] _{AN}
Blue Chalk					
1976-77	4.5	8.1	— ¹	7.8	7.5
1977-78	6.5	7.9	6.6	7.7	7.7
1978-79	6.4	7.0	8.1	7.0	6.8
1979-80	6.9	6.5	7.6	6.5	6.3
1976-80	6.1	7.4	7.4	7.3	7.1
Chub					
1976-77	13.3	11.7	11.6	11.9	11.9
1977-78	13.5	12.2	— ¹	12.4	12.4
1978-79	11.7	12.5	15.1	12.5	12.6
1979-80	15.2	11.6	9.4	11.8	11.8
1976-80	13.4	12.0	12.0	12.2	12.2
Dickie					
1976-77	12.8	10.0	13.2	12.7	13.1
1977-78	13.4	14.4	11.5	13.2	11.8
1978-79	12.4	13.1	14.0	13.2	13.4
1979-80	16.9	14.8	10.7	14.6	13.1
1976-80	13.4	13.1	12.3	13.4	12.8
Harp					
1976-77	7.3	8.1	10.5	8.1	8.0
1977-78	6.7	8.4	9.2	8.2	8.1
1978-79	8.6	7.2	8.7	7.5	7.4
1979-80	8.5	7.5	6.8	7.5	7.7
1976-80	7.8	7.8	8.8	7.8	7.8
Jerry					
1976-77	— ¹	10.5	9.9	10.5	9.8
1977-78	7.7	9.9	9.7	9.7	9.5
1978-79	6.9	8.5	8.0	8.3	8.5
1979-80	8.6	8.8	9.5	8.9	8.8
1976-80	7.7	9.4	9.3	9.3	9.1
Red Chalk					
1976-77	5.0	5.3	6.4	5.4	5.7
1977-78	6.8	6.4	5.1	6.4	6.1
1978-79	4.7	5.4	7.4	5.5	5.6
1979-80	5.4	5.1	4.8	5.3	5.4
1976-80	5.5	5.6	5.9	5.6	5.7

¹ no data

Table 47. TP concentration at spring overturn from 1980 to 1982. All results in mg m^{-3} .

Lake	1980	1981	1982
Blue Chalk	6.5	6.3	6.7
Chub	12.6	12.4	10.6
Dickie	11.5	12.0	9.5
Harp	8.3	7.0	7.0
Jerry	8.1	— ¹	— ¹
Red Chalk	4.8	4.6	4.9

¹ no data

ii) IMPROVEMENTS IN TP CONCENTRATION MODELS

Many alterations to the simple mass balance model have been proposed in the past decade. Changes can be classified as being of two general types:

a) those that are based on a more "realistic" concept of a lake. (Figure 11b). For example, some of these consider thermal stratification (resulting in a 2- or 3-layer model for part of the year), TP release from the sediments to the water column, specific fractions of the TP pool (e.g. particulate phosphorus and dissolved phosphorus), and other modifications;

b) those that evaluate model parameters and coefficients by statistical techniques such that the resulting model is empirical or semi-empirical rather than a true mass balance model (eg. Reckhow 1977, Walker 1977).

Some of these changes are considered in this section but the approach taken is to use the simplest model that makes accurate predictions. This approach was followed principally because the simplest models generally require the least measurements. Improvements based on a more "realistic" conceptual model of a lake are discussed first.

In instances where input parameters are expressed on an annual basis, the time period for which the prediction of TP concentration is applied has been considered to be the spring overturn in some cases (Dillon and Rigler 1974a), the annual average in others and the ice-free period average in yet other instances. Although empirical models may be tested or calibrated on any temporal basis, mass balance models which are semi-empirical (i.e. contain some empirical and some non-empirical steps) or non-empirical may best predict annual TP concentrations or at least TP concentrations representing most of the year. Since it is desired to predict trophic status parameters (chlorophyll *a*, phytoplankton biomass, Secchi disc depth) during the ice-free period on the basis of TP concentration, the model was applied to predict mean ice-free values of TP concentration. The difficulty of obtaining good estimates of mean annual TP concentration, because of the problems encountered in sampling in winter, reinforces the decision to use averages for the ice-free period. To investigate further the consequences of this decision, the differences in TP concentration between seasonal time periods in the study lakes were tested (data summarized in Table 46). There were no significant differences (paired t-test, $p > 0.05$) between the mean TP concentrations of the ice-free period and those of the spring overturn, summer stratified, fall overturn or annual periods, nor were there any significant differences between values measured in the spring overturn and any other time period. These results suggest that seasonal patterns in the TP concentrations in the study lakes were not pronounced. However, highly significant correlations between the TP concentrations measured in different time periods were found. The relationships between TP concentrations in the ice-free period and those in four other time periods (spring overturn, summer stratified, fall overturn, annual) are given in Table 48. These relationships allow estimates of TP concentration for these periods to be made from the predictions made by the ice-

free model, or estimates of ice-free concentration to be made from measurements made during other time periods.

Differences in TP concentration in vertical depth zones, resulting from thermal stratification were observed in some study lakes (Table 49). The TP concentrations in the hypolimnia of Blue Chalk, Chub and Dickie Lakes (in years when it was thermally stratified) were much greater than concentrations measured in the epilimnia of the same lakes (paired t-test, $p < 0.05$), with differences of 5.9-8.1 mg m^{-3} . The differences in Harp, Jerry and the main basin of Red Chalk Lake were much less, ranging from only 0.6-3.2 mg m^{-3} . Mass balance models incorporating thermal stratification during part of the year have been proposed and utilized (Snodgrass and O'Melia 1975). Snodgrass and Dillon (1983) compared the predictive abilities of one and two-layer models using a data set collected in the Muskoka-Haliburton study area (Dillon 1974). The one-layer model, similar in structure to that proposed in equation 15, provided equally accurate predictions of TP concentrations as did the two-layer model (an epilimnion and a hypolimnion with the thermocline treated as a two-dimensional plane). Therefore, it is concluded that it is unnecessary to modify the mass balance model to account for thermal stratification.

Table 48. Relationships between TP concentration (mg m^{-3}) in the ice-free period and that measured in 4 other time periods. Results for each time period were averaged over the 4 years for each lake.

A. Predicting ice-free period concentration

ice-free vs. spring overturn	
$[\text{TP}]_{\text{IF}} = 0.80 [\text{TP}]_{\text{SO}} + 2.04$	$r^2 = 0.943$
ice-free vs. fall overturn	
$[\text{TP}]_{\text{IF}} = 1.04 [\text{TP}]_{\text{FO}} - 0.15$	$r^2 = 0.913$
ice-free vs. summer stratified	
$[\text{TP}]_{\text{IF}} = 1.04 [\text{TP}]_{\text{SS}} - 0.33$	$r^2 = 0.998$
ice-free vs. annual	
$[\text{TP}]_{\text{IF}} = 1.04 [\text{TP}]_{\text{AN}} - 0.23$	$r^2 = 0.996$

B. Predicting other concentrations from ice-free concentration

spring overturn vs. ice-free	
$[\text{TP}]_{\text{SO}} = 1.18 [\text{TP}]_{\text{IF}} - 1.91$	$r^2 = 0.943$
fall overturn vs. ice-free	
$[\text{TP}]_{\text{FO}} = 0.88 [\text{TP}]_{\text{IF}} + 0.91$	$r^2 = 0.913$
summer stratified vs. ice-free	
$[\text{TP}]_{\text{SS}} = 0.96 [\text{TP}]_{\text{IF}} + 0.33$	$r^2 = 0.998$
annual vs. ice-free	
$[\text{TP}]_{\text{AN}} = 0.96 [\text{TP}]_{\text{IF}} + 0.25$	$r^2 = 0.996$
epilimnetic vs. ice-free	
$[\text{TP}]_{\text{EP}} = 0.95 [\text{TP}]_{\text{IF}} - 1.23$	$r^2 = 0.903$
epi- and metalimnetic vs. ice-free	
$[\text{TP}]_{\text{EM}} = 0.90 [\text{TP}]_{\text{IF}} + 0.09$	$r^2 = 0.884$

C. Predictions based on summer stratification concentration

epilimnetic vs. summer stratification	
$[\text{TP}]_{\text{EP}} = 0.98 [\text{TP}]_{\text{SS}} - 1.44$	$r^2 = 0.881$
epi- and metalimnetic vs. summer stratification	
$[\text{TP}]_{\text{EM}} = 0.93 [\text{TP}]_{\text{SS}} - 0.11$	$r^2 = 0.863$

Table 49. Total phosphorus concentration (mg m^{-3}) in the epilimnion (subscript E), epi- and metalimnion (subscript EM), hypolimnion (subscript H) and in the whole lake during the summer stratification (subscript SS) period for "A" lakes for each of 4 12-month periods and for 1976-80. Concentrations in the epi- and metalimnion and during summer stratification are volume-weighted averages.

Lake/year	$[\text{TP}]_{\text{E}}$	$[\text{TP}]_{\text{EM}}$	$[\text{TP}]_{\text{H}}$	$[\text{TP}]_{\text{SS}}$
Blue Chalk				
1976-77	5.6	7.1	14.1	8.1
1977-78	6.3	6.9	16.4	7.9
1978-79	5.6	6.3	12.6	7.0
1979-80	5.1	5.7	11.7	6.5
1976-80	5.7	6.5	13.7	7.4
Chub				
1976-77	9.1	9.3	16.5	11.7
1977-78	8.9	9.4	19.6	12.2
1978-79	10.6	10.7	16.9	12.5
1979-80	8.7	9.0	16.8	11.6
1976-80	9.3	9.6	17.4	12.0
Dickie				
1976-77	13.5	14.1	— ¹	10.0
1977-78	11.4	13.1	17.8	14.4
1978-79	12.1	12.7	17.9	13.1
1979-80	13.7	14.2	20.0	14.8
1976-80	12.7	13.5	18.6	13.1
Harp				
1976-77	7.3	7.9	8.5	8.1
1977-78	7.5	8.2	8.9	8.4
1978-79	6.2	7.0	7.9	7.2
1979-80	6.4	7.3	8.0	7.5
1976-80	6.9	7.6	8.3	7.8
Jerry				
1976-77	6.5	8.3	13.6	10.5
1977-78	7.5	8.5	12.5	9.9
1978-79	7.3	8.0	9.8	8.5
1979-80	7.8	8.0	10.3	8.8
1976-80	7.3	8.2	9.3	9.4
Red Chalk				
1976-77	4.5	5.6	5.2	5.3
1977-78	5.5	6.6	6.3	6.4
1978-79	5.0	5.7	5.1	5.4
1979-80	4.4	5.1	5.2	5.1
1976-80	4.9	5.7	5.5	5.6

¹ no hypolimnion

The assumption that the TP concentration in the lake outlet is equal to the concentration in the lake is inherent in the simple mass balance model. Based on earlier published studies, Dillon et al. (1978) suggested that the relationship in TP concentration between lake outlet and lake concentrations could be approximated as:

$$[\text{TP}]_{\text{OUT}} = 0.9 [\text{TP}]_{\text{AN}} \quad (25)$$

where: $[\text{TP}]_{\text{AN}}$ = mean annual TP concentration in the lake.

The mean annual TP concentrations of the outflows of the study lakes are reported in Table 50. The relationship between these concentrations and the mean ice-free period lake concentrations is:

$$[\text{TP}]_{\text{OUT}} = 0.956 [\text{TP}]_{\text{IF}} \quad (r^2 = 0.80) \quad (26)$$

If this relationship is considered in the model derivation described in equations 17 and 24, the resulting solution is:

$$[\text{TP}]^1 = L(1 - R_p) / 0.956 q_s \quad (27)$$

$$\text{or } [\text{TP}]^1 = L / (0.956 q_s + 12.4) \quad (28)$$

In subsequent calculations, the model defined by equation 28 is termed "LCS-1".

An alternate method of deriving the loss of TP in a lake by sedimentation was outlined in Section III.B.8. Site-specific whole-lake Pb burdens were used to pro-rate measured sediment accumulation rates (Pb-210 methodology), which were combined with the measured TP content of the sediments to estimate the long-term whole-lake TP sedimentation rate (S).

The mass balance model can therefore be re-formulated as:

$$[\text{TP}]^1 = (L - S) / 0.956 q_s \quad (29)$$

This model is subsequently described as "LCS-2".

Other advances in TP model development have been based on statistical techniques. Model coefficients have been empirically derived by fitting measured data to the basic mass balance model. For example, Reckhow (1977) used a global data set to develop the relationship:

$$[\text{TP}] = L / (11.6 + 1.2 q_s) \quad (30)$$

where the 11.6 (equivalent to the settling velocity) and the 1.2 were estimated using statistical optimization techniques. Using the same techniques but an expanded data set, Walker (1977) proposed an alternative model:

$$[\text{TP}] = L / q_s (1 + 0.824 \tau_w^{0.454}) \quad (31)$$

Table 50. Mean annual TP concentration (mg m^{-3}) in the outflow of the study lakes.

Lake	1976-77	1977-78	1978-79	1979-80
Blue Chalk	7.33	9.06	9.86	8.15
Chub	11.1	10.7	9.77	10.5
Dickie	12.0	13.0	12.5	12.4
Harp	9.36	7.52	7.88	8.93
Jerry	8.27	8.48	7.63	9.53
Red Chalk	9.42	7.04	5.99	7.08

Table 51. Empirically derived predictive mass balance models. These models were developed using statistical techniques and are formulated on the basis of best fit to global data sets.

Source	Model	Conditions
Reckhow (1977)	$[\text{TP}] = L / (11.6 + 1.2 q_s)$	general
Walker (1977)	$[\text{TP}] = L / q_s (1 + 0.824 \tau_w^{0.454})$	general
Reckhow (1977)	$[\text{TP}] = L / ((18\bar{z} / (10 + \bar{z}) + 1.05 q_s \exp(0.012 q_s)))$	oxic lakes, $q_s < 50 \text{ m yr}^{-1}$
Reckhow (1977)	$[\text{TP}] = L / (0.17\bar{z} + 1.13 q_s)$	anoxic lakes

Significant improvements in predictive capability were achieved by Reckhow (1977), when he divided the lake set into those with completely oxic hypolimnia throughout the period of stratification and those with partially or totally anoxic hypolimnia. Only those lakes with $q_s < 50 \text{ m yr}^{-1}$ were considered in the oxic subset in his analysis.

These results (summarized in Table 51) indicate that there are inherent differences in the way in which lakes with anoxic as opposed to oxic hypolimnia process TP. This finding is consistent with the observation that the study lakes with anoxic hypolimnia had greater differences between hypolimnetic and epilimnetic TP concentration than did those that had oxic hypolimnia. These individual models were also tested using the data collected for the study lakes.

iii) MODEL CALIBRATION - JERRY LAKE

The previous sections outlined the rationale for use of the TP mass balance lake models in a form based on the following assumptions:

- the TP concentration of the lake is in steady-state,
- the model predicts the mean TP concentration in the ice-free period, but it can be related to the TP concentration in other periods,
- the input data to the model are averages of four years (1976-1980) of data,
- the TP inputs to the lake are mixed throughout the water column at all times; i.e. the one-layer version of the model (no thermal stratification) is suitable,
- the concentration of TP in the lake outlet is related to the concentration of TP in the lake by the relationship shown in equation 26.

There was no potential contribution of TP from shoreline development to Jerry Lake (Table A18), and the potential input to Red Chalk Lake (Table A17) was close enough to zero 50 as to be insignificant (potential anthropogenic input = 1.1% total natural input). Therefore, both lakes can be used to validate the TP mass balance lake model.

Except for the sedimentation rate (or retention coefficient), the information needed to predict the TP concentration of Jerry Lake using the model designated as LCS-1 is provided in Tables 14 and A18. Since $J_A = 0$ and since the TP concentration is in steady-state, the measured retention represents the actual loss of TP to the lake's sediments. However, this value of retention cannot be used for prediction purposes since it is not measured independently of the TP budget.

The retention coefficients predicted on the basis of the model outlined in equation 22 (with settling velocity $v = 12.4 \text{ m yr}^{-1}$) are compared to measured retention coefficients in Table 52. As expected, the long-term (1976-1980) average measured and predicted retention coefficients are in much better agreement than the individual annual values. This occurs because the lake response times are > 1 year and changes in annual input of TP are

Table 52. Measured (by mass balances) and predicted TP retention (R_p) in Jerry Lake. Predicted retention calculated using settling velocity of 12.4 m yr^{-1} .

	R_p				
	1976-77	1977-78	1978-79	1979-80	1976-80
Measured	0.734	0.652	0.551	0.420	0.598
Predicted	0.634	0.618	0.529	0.582	0.588

therefore not necessarily reflected in corresponding changes in output (and therefore retention) in the same year. If the response time of the lake was rapid (<1 year), then better agreement on an annual basis would be expected. Conversely, lakes with water replenishment rates of decades would require even longer-term data than those collected in this study.

Since the prediction of TP concentration using the mass balance model (equation 27) is directly proportional to the term $(1-R_p)$, comparison of predicted and measured retention is, in effect, a test of the predictive ability of the mass balance model. Nevertheless, the TP mass balance model can be validated in a more conventional manner by comparing predicted vs observed concentrations of TP. The calculations of the predicted TP concentration are shown in Table 53 using a value of retention predicted on the basis of a settling velocity of 12.4 m yr^{-1} . The predicted steady-state TP concentration (9.3 mg m^{-3}) is identical to the average concentration measured in the ice-free period (Table 46). Therefore, the observed TP concentration can be adequately explained by the measured TP input and that the retention model is satisfactory.

Table 53. Prediction of TP concentration in Jerry Lake using LCS-1 model (equations 27 and 32)

Parameter	Source	4-year average
L	Table 61	$187.5 \text{ mg m}^{-2} \text{ yr}^{-1}$
q_s	Table 20	8.69 m yr^{-1}
R_p	Table 70	0.588
[TP]	$\frac{L(1-R_p)}{0.956 q_s}$	9.3 mg m^{-3}
[TP] _{MEAS}	Table 64	9.3 mg m^{-3}

An alternative calculation (LCS-2) can be made using the estimated long-term accumulation of TP in the sediments of Jerry Lake ($99 \text{ mg TP m}^{-2} \text{ yr}^{-1}$ calculated from the Pb-210 method (Table 40)) to calculate the retention coefficient. This method yields a prediction of $[TP] = 10.6 \text{ mg m}^{-3}$, somewhat higher than the measured $[TP]$ of 9.3 mg m^{-3} .

iv) MODEL CALIBRATION - RED CHALK LAKE

As previously discussed, Red Chalk Lake data can be used as an additional validation of the TP mass balance model because the potential anthropogenic TP input is insignificant. If the measured and predicted (using settling velocity $v = 12.4 \text{ m yr}^{-1}$) TP retention coefficients are compared (Table 54) based on 1976-1980 data, it is apparent that the predicted retention is greater than the measured. There is substantial variability between individual years, with the measured retention exceeding the predicted in one of the four years.

Calculations based on the mass balance model (Table 55) yield a predicted TP concentration of 5.4 mg m^{-3} , which is within experimental error of the measured (Table 46) value¹ of 5.6 mg m^{-3} . Despite this good agreement, it was believed that the small eastern basin of Red Chalk Lake may act as a disproportionately important sink for TP

since most of the TP input passes into this basin before the main basin. This hypothesis is supported by the observation (Table 56) that the TP concentration in the east basin was substantially greater than that in the main basin during the ice-free period (7.8 mg m^{-3} vs 5.6 mg m^{-3}). Further comparisons of the TP concentrations in the east basin and main basin at spring overturn (7.4 mg m^{-3} vs 5.5 mg m^{-3} respectively), fall overturn (6.3 mg m^{-3} vs 5.9 mg m^{-3} respectively) and at other time periods demonstrated that the TP concentration of the east basin was always greater. However, by far the greatest differential in TP concentrations between the two basins was in the hypolimnetic TP values (18.0 mg m^{-3} vs 5.5 mg m^{-3} respectively).

Table 54. Measured (by mass balances) and predicted TP retention in Chalk Lake. Predicted retention calculated using settling velocity of 12.4 m yr^{-1} .

	R_p				
	1976-77	1977-78	1978-79	1979-80	1976-80
Measured	0.789	0.630	0.516	0.470	0.608
Predicted	0.739	0.706	0.636	0.640	0.677

Table 55. Prediction of TP concentration in Red Chalk Lake using LCS-1 model (equations 27 and 32)

Parameter	Source	4-year average
L	Table 61	$94.2 \text{ mg m}^{-2} \text{ yr}^{-1}$
q_s	Table 20	5.91 m yr^{-1}
R_p	Table 72	0.677
[TP]	$\frac{L(1-R_p)}{0.956 q_s}$	5.4 mg m^{-3}
[TP] _{MEAS}	Table 64	5.6 mg m^{-3} (5.8) ¹

¹ see footnote on page

In order to clarify the causes of these differences, hydrologic and TP mass balances were constructed separately for each basin. The transfer of water from the east basin to the main basin was calculated as the residual term in the water balance (all other terms measured) for Red Chalk East (Table 57) since no direct measurements were available. This estimate was corrected to account for the error observed in the Red Chalk (whole lake) water balance (Table A6) by apportioning the error term on the basis of the ratio of the watershed area of Red Chalk East to Red Chalk (whole basin). The areal water load (q_s) was calculated for Red Chalk East as the estimated outlet Q /lake area of Red Chalk East. Similarly, the areal water load of the main basin was calculated as outlet Q /area main basin (Table 58). A measure of the transport of TP from the east basin to the main basin was derived using the estimated outflow from the east basin and the TP concentration in the basin. Using this information and

¹ The mean TP concentration of 5.6 mg m^{-3} was calculated from measurements taken at the main basin of Red Chalk Lake. Calculating the mean using measurements taken at both the main and east basins (weighted to account for differences in volume of water between the basins), a value of 5.8 mg m^{-3} is obtained.

Table 56. Comparison of total phosphorus concentrations (mg m^{-3}) in the main basin and east basin of Red Chalk Lake. Four annual averages and the 4-year average are reported for different seasons and different depth zones (defined in Tables 46 and 49).

	[TP] _{SO}	[TP] _E	[TP] _{EM}	[TP] _H	[TP] _{SS}	[TP] _{IF}	[TP] _{FO}	[TP] _{AN}
Main basin								
1976-77	5.0	4.5	5.6	5.2	5.3	5.4	6.4	5.7
1977-78	6.8	5.5	6.6	6.3	6.4	6.4	5.1	6.1
1978-79	4.7	5.0	5.7	5.1	5.4	5.5	7.4	5.6
1979-80	5.4	4.4	5.1	5.2	5.1	5.3	4.8	5.4
1976-80	5.5	4.9	5.7	5.5	5.6	5.6	5.9	5.7
East basin								
1976-77	—	—	—	—	—	—	—	—
1977-78	7.5	5.5	7.4	20.5	8.7	8.6	7.4	8.5
1978-79	6.7	5.0	6.1	16.2	7.5	7.4	5.5	7.2
1979-80	8.1	4.7	5.8	17.1	7.4	7.3	6.0	7.1
1976-80	7.4	5.1	6.5	18.0	7.9	7.8	6.3	7.6

other data from Table A17, the TP mass balances for the east basin and main basin of Red Chalk Lake were constructed (Tables 59 and 60 respectively).

The predicted concentration of TP in each of the basins is shown in Table 61. In the main basin, the predicted and observed concentrations (ice-free period) were almost identical (5.5 mg m^{-3} vs 5.6 mg m^{-3} respectively), whereas in the east basin the predicted concentration was substantially lower than the observed (6.4 mg m^{-3} vs 7.8 mg m^{-3} respectively). The TP mass balance model therefore provided accurate predictions of TP concentration in Jerry and Red Chalk (Main basin) Lakes but not in Red Chalk East. The difference between these lakes was that the hypolimnia of Jerry and Red Chalk Main were oxic throughout the period of summer stratification while the hypolimnion of Red Chalk East was not (Figure 12). A manifestation of this was that the TP concentration in the hypolimnion of the east basin of Red Chalk Lake was much greater than that in the epilimnion (Table 56) whereas this was not the case in the main basin of Red Chalk Lake or in Jerry Lake (Table 49).

Table 57. Water balance for Red Chalk Lake East Basin. All results in $10^6 \text{ m}^3 \text{ yr}^{-1}$ with the exception of q_s (m yr^{-1}) and R_p (calculated using equation 22).

Source	1976-77	1977-78	1978-79	1979-80	mean
INPUTS					
Blue Chalk Outflow	0.644	0.713	0.962	1.030	0.837
Inflow 1	0.648	0.695	0.872	0.885	0.775
Inflow 2	0.094	0.120	0.175	0.149	0.135
Precipitation	0.107	0.106	0.176	0.153	0.136
Ungaaged	0.108	0.140	0.170	0.160	0.145
Total	1.601	1.774	2.355	2.377	2.027
OUTPUTS					
Evaporation	0.093	0.075	0.089	0.085	0.086
Storage	0.000	+0.015	+0.004	-0.003	+0.004
Outflow	1.508	1.684	2.262	2.295	1.937
Corrected outflow ¹	1.651	1.814	2.491	2.527	2.121
q_s	12.7	14.0	19.2	19.4	16.3
R_p	0.494	0.470	0.392	0.390	0.432

¹ error in the whole-lake water balance divided between the East and Main Basins on the basis of relative watershed and basin areas.

Table 58. Water balance for Red Chalk Lake Main Basin. Outflow is expressed in $10^6 \text{ m}^3 \text{ yr}^{-1}$; q_s in m yr^{-1} . R_p is calculated from equation 22.

Source	1976-77	1977-78	1978-79	1979-80	mean
Outflow	2.490	2.940	4.040	3.970	3.360
q_s	5.67	6.70	9.20	9.04	7.65
R_p	0.686	0.649	0.574	0.578	0.622

Table 59. Total phosphorus mass balance for Red Chalk Lake East Basin. All results in kg yr^{-1} , except load ($\text{mg m}^{-2} \text{ yr}^{-1}$).

Source	1976-77	1977-78	1978-79	1979-80	mean
Blue Chalk Outflow	3.49	4.86	5.57	6.51	5.11
Inflow 1	6.23	9.32	9.58	12.50	9.41
Inflow 2	1.31	1.69	2.01	1.67	1.67
Precipitation	7.59	4.73	3.93	2.83	4.77
Ungaaged	1.46	2.02	1.83	2.01	1.83
Total	20.08	22.62	22.92	25.52	22.79
Load	154.5	174.0	176.3	196.3	175.3

Table 60. Total phosphorus mass balance for Red Chalk Lake Main Basin. All results in kg yr^{-1} , except load ($\text{mg m}^{-2} \text{ yr}^{-1}$).

Source	1976-77	1977-78	1978-79	1979-80	mean
Red Chalk East	12.88	15.60	18.43	18.45	16.34
Inflow 3	4.31	7.83	6.44	6.20	6.20
Inflow 4	5.32	6.41	5.19	4.93	5.46
Precipitation	25.61	15.97	13.27	9.57	16.11
Ungaaged	2.42	3.33	3.01	3.31	3.02
Total	50.54	49.14	46.34	42.46	47.12
Load	115.1	111.9	105.6	96.7	107.3

As mentioned previously in this section, Reckhow (1977) proposed that different mass balance models were required for oxic and anoxic lakes. In Table 62, predictions of TP concentration made on the basis of Reckhow's oxic and anoxic models, the Reckhow general model, the Walker (1977) general model and the LCS-1 and LCS-2 models (equations 28 and 29) derived in this study are compared to the measured TP concentrations in Jerry

Lake and the two basins (plus the whole lake) of Red Chalk Lake. For the oxic lakes, the LCS-1 model makes the best predictions, although the Reckhow oxic model is almost as good. The Reckhow general model, which makes no distinction between oxic and anoxic lakes, underestimated the TP concentrations, whereas the Walker model overestimated them. The LCS-2 model made poorer predictions than LCS-1, and was excluded from subsequent calculations. However, none of the models provided an accurate prediction of the TP concentration of the anoxic east basin of Red Chalk Lake.

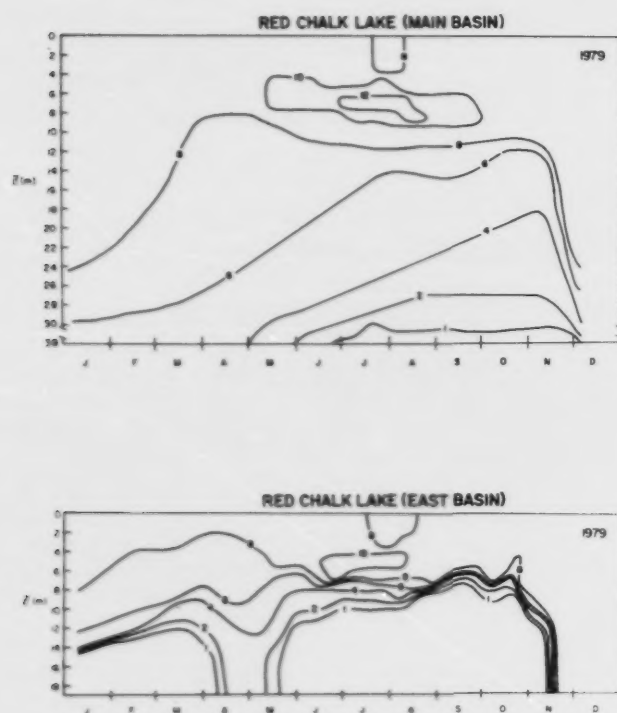


Figure 12 Dissolved oxygen (mg L^{-1}) isopleth maps for Red Chalk Main and Red Chalk East Basin for 1979.

On the basis of these findings, it was decided that an alternate formulation of the mass balance model for anoxic lakes was required. Increased TP concentration in the hypolimnion of the east basin indicates either release of TP from the sediments or less efficient removal from the water column. These two possibilities are indistinguishable, and can be reported mathematically by a decrease in the settling velocity. Since the anthropogenic input of TP from lakeshore development can make no contribution to the TP mass balance or the TP concentration of the east basin of Red Chalk Lake, the LCS-1 model can simply be calibrated by optimizing the settling velocity (v) such that the predicted and observed TP concentrations agree. The result of this procedure is that for anoxic lakes the proposed settling velocity is 7.2 m yr^{-1} , as opposed to 12.4 m yr^{-1} for oxic lakes. The LCS model therefore has the form:

$$[\text{TP}]^1 = L(1 - R_p) / 0.956 q_s \quad (27)$$

where: $R_p = 12.4 / (12.4 + q_s)$ for oxic lakes (32)

$R_p = 7.2 / (7.2 + q_s)$ for anoxic lakes (33)

Table 61. Prediction of TP concentration (mg m^{-3}) in each basin of Red Chalk Lake.

Parameter	Main basin		East basin	
	Source	4-year average	Source	4-year average
$L (\text{mg m}^{-2} \text{ yr}^{-1})$	Table 78	107.3	Table 77	175.3
$q (\text{m yr}^{-1})$	Table 76	7.65	Table 75	16.3
R_p	Table 76	0.622	Table 75	0.432
$[\text{TP}] (\text{mg m}^{-3})$	$\frac{L(1-R_p)}{0.956q_s}$	5.5(4)	$\frac{L(1-R_p)}{0.956q_s}$	6.4
$[\text{TP}]_{\text{MEAS}} (\text{mg m}^{-3})$	Table 74	5.6	Table 74	7.8

Table 62. Comparison of predicted and measured TP concentrations (mg m^{-3}) for Jerry and Red Chalk Lakes. Predictions are based on 4 models reported in the literature and the LCS models resulting from this study.

	Jerry	Red Chalk		
		Whole	Main	East
PREDICTED				
Reckhow – general	8.5	5.0	5.2	5.6
Reckhow – oxic	9.3	5.5	5.3	–
Reckhow – anoxic	–	10.4	–	9.0
Walker – general	11.0	7.2	6.5	7.1
LCS – 1	9.3	5.4	5.5(4)	6.4
LCS – 2	10.6	1.6	2.3	6.2
MEASURED				
	9.3	5.6 ¹	5.6	7.8

¹ measured in Main basin.

v) EFFECTS OF LAKESHORE DEVELOPMENT

The potential contribution of anthropogenic TP from lakeshore development to the TP mass balances and TP concentrations of Chub, Blue Chalk, Dickie and Harp Lake is significant (Table 45). In order to estimate the proportion of this potential TP input that actually entered the lake, the model developed and calibrated in the previous section was employed.

The potential contribution of anthropogenic TP to the TP supply of Chub and Blue Chalk Lake was originally estimated to be 26% and 27% of the total input respectively (Table 1), based on estimated natural inputs and number of dwelling units. However, measurements made as part of this study (Tables 32, A15, A17) indicated that the revised potential contribution of anthropogenic TP was only 7% and 19% of the total for Chub and Blue Chalk Lakes respectively (Table 45). The effect of this input on Chub Lake could only be small.

Both Chub and Blue Chalk Lake had anoxic hypolimnetic waters during late summer and in the fall prior to destratification (Reid et al. 1983a). Therefore the anoxic models (equations 27 and 33) are suitable for these lakes. The TP concentration predicted on the basis of the measured natural TP inputs was less than the measured concentration (Table 63) in both lakes. The difference in Chub Lake was 1.8 mg m^{-3} whereas it was 1.5 mg m^{-3} in Blue Chalk Lake. Although these differences are not great, they are both in excess of the estimated analytical errors

in the measurements (0.5 mg m^{-3}) and much greater than the differences between predicted and observed TP concentrations in the case of Red Chalk Lake (Main basin)¹ and Jerry Lake (predicted TP concentration agreed with observed to within 0.1 mg m^{-3} in both lakes). If the entire potential contribution of anthropogenic TP from lakeshore development (L_A in Table 63) is included in the TP mass balance, then the predicted and observed TP concentrations in the lakes are in much better agreement, differing by 1.0 mg m^{-3} in Chub Lake and 0.1 mg m^{-3} in Blue Chalk Lake. In each case, the measured TP concentration is still greater than the predicted concentration, although the difference for Blue Chalk Lake is insignificant. The results indicate that shoreline development is contributing to the TP concentration in these lakes. In fact, the results suggested that additional inputs of TP to Chub Lake occurred. Although measurement or model error could explain this underestimate, it is also possible that the data on human use (capita years yr^{-1}) of the lake are in error. Survey information (Table 31) suggested human use of Chub Lake to be 4.01 capita years yr^{-1} in 1976. Estimating human use by the "accessibility equation" (Downing 1983), the "cottager equation" (Downing 1983) and the "average equation" (Downing 1983, using Blue Chalk Lake as a benchmark lake) gives values of 8.4 cap yrs yr^{-1} , 6.6 cap yrs yr^{-1} and 6.4 cap yrs yr^{-1} respectively. Estimates of human use for Chub Lake in 1978 obtained by employing the latter three methods (9.2, 7.3 and 7.1 cap yrs yr^{-1}) were also considerably higher than the estimate obtained from survey information (4.4 cap yrs yr^{-1}). If the average value of human use calculated from the three equations outlined in Downing (1983) is used, a potential anthropogenic TP input of $15.1 \text{ mg m}^{-2} \text{ yr}^{-1}$ is obtained. This, in turn, results in a predicted TP concentration of 11.8 mg m^{-3} , in close agreement with the observed value of 12.2 mg m^{-3} .

Original estimates (Table 1) indicated that the potential contributions of anthropogenic TP from shoreline development to the TP mass balances of Harp Lake and Dickie Lake were 61% and 54% respectively. Although these figures were reduced (to 55% and 46% respectively; Table 45) when measured data on natural inputs and

Table 63. Areal water load (q_s ; m yr^{-1}), TP retention coefficient (R_p ; from equation 33), measured natural TP load (L_N ; $\text{mg m}^{-2} \text{ yr}^{-1}$), maximum possible TP load from shoreline development (L_A ; $\text{mg m}^{-2} \text{ yr}^{-1}$), maximum total load ($L_T = L_A + L_N$; $\text{mg m}^{-2} \text{ yr}^{-1}$), measured TP concentration in the lake ($[TP]_{\text{MEAS}}$; mg m^{-3}), and predicted TP concentration based on natural load ($[TP]_N$) and total load ($[TP]_T$) using anoxic model (equations 27 and 33) for Chub and Blue Chalk Lakes.

	Chub	Blue Chalk
q_s	4.40	1.69
R_p	0.621	0.810
L_N	115.5	49.3
L_A	8.5	11.5
L_T	124.0	60.8
$[TP]_{\text{MEAS}}$	12.2	7.3
$[TP]_N$	10.4	5.8
$[TP]_T$	11.2	7.2

human use were collected, the potential contribution of the anthropogenic TP remained high.

The hypolimnion of Harp Lake remained oxygenated at all times allowing use of the oxic model (equations 27 and 32). However, Dickie Lake did not have a hypolimnion in all years of study (Reid et al. 1983a), undoubtedly because of its small maximum depth (12m) and relatively large surface area ($\approx 0.93 \text{ km}^2$). Dickie Lake was only weakly stratified in all years, with two discernible layers in the summer of 1976 and three in the subsequent three summers. The oxygen content of the bottom waters was low ($< 1 \text{ mg L}^{-1}$) in each of the years, but the bottom waters remained oxic (Figure 13). However, in two of the three subsequent summers (1981, 1982), the bottom waters were anoxic. The calculations were therefore carried out for Dickie Lake using both the anoxic and oxic models, although the information gathered indicates that the oxic model was more appropriate.

The TP concentration predicted for Harp Lake using the measured natural load ($101 \text{ mg TP m}^{-2} \text{ yr}^{-1}$) was less than the observed concentration (6.4 vs 7.8 mg m^{-3} ; Table 64). However, if all of the potential anthropogenic TP input from shoreline development reached the lake, the expected concentration (12.8 mg m^{-3}) exceeded the measured concentration by a considerable margin. The mass balance model can be used to estimate the anthropogenic TP input by equating the observed TP concentration with the model formulation. If this is done, the total load must have been $123.8 \text{ mg TP m}^{-2} \text{ yr}^{-1}$. Therefore, the actual input from shoreline development ($22.8 \text{ mg m}^{-2} \text{ yr}^{-1}$) represents 19% of the potential contribution.

The predicted TP concentration in Dickie Lake based on the oxic model and the natural input, was much less than the measured concentration (6.7 vs 13.4 mg m^{-3}). If all of the potential input from shoreline development was included, the predicted concentration (12.5 mg m^{-3}) was slightly less than that measured. In the less likely case that the anoxic model is more suitable for Dickie Lake, the predicted and measured concentrations can be compared

Table 64. Areal water load (q_s ; m yr^{-1}), TP retention coefficient (R_p ; from equation 32), measured natural TP load (L_N ; $\text{mg m}^{-2} \text{ yr}^{-1}$), maximum possible TP load from shoreline development (L_A ; $\text{mg m}^{-2} \text{ yr}^{-1}$), maximum total load (L_T ; $\text{mg m}^{-2} \text{ yr}^{-1}$), measured TP concentration in the lake ($[TP]_{\text{MEAS}}$; mg m^{-3}), and predicted TP concentration based on natural load ($[TP]_N$) and total load ($[TP]_T$) using oxic model (equations 27 and 32) for Harp and Dickie Lakes.

	Harp	Dickie
q_s	4.20	2.60
R_p	0.747	0.827 (0.735)
L_N	101.0	96.1
L_A	121.8	83.3
L_T	222.8	179.4
$[TP]_{\text{MEAS}}$	7.8	13.4
$[TP]_N$	6.4	6.7 (10.2)
$[TP]_T$	12.8	12.5 (19.1)

¹ The east basin of Red Chalk Lake was used for model calibration and therefore cannot be used to provide an error estimate.

in a similar manner. With no anthropogenic input, the prediction underestimates the measured concentration (10.2 vs 13.4 mg m^{-3}); while with 100% of the anthropogenic input, the prediction (19.1 of mg m^{-3}) overestimates the measured concentration. Calculation of the anthropogenic load from the observed concentration yields $29.4 \text{ mg TP m}^{-2} \text{ yr}^{-1}$, which is 36% of the potential input.

To briefly recapitulate the principal findings of this section of the report:

i) A simple mass balance model to predict the TP concentration in a lake was described. This model was modified to suit conditions in the study lakes and may be described mathematically as follows:

$$[\text{TP}]^i = L(1 - R_p) / 0.956 q_s$$

ii) The model was tested on undeveloped Jerry and Red Chalk Lake (Main basin). The predicted and observed TP concentrations agreed to within 0.1 mg m^{-3} , giving confidence in the use of the model for developed lakes.

iii) The settling velocity (v) of TP in the lake was found to differ in lakes with oxic and anoxic hypolimnia. As the value of v affects the prediction of R_p in the model, this resulted in different model formulations for these two types of lakes. The value of v for a lake with an anoxic hypolimnion was set by calibration on the undeveloped east basin of Red Chalk Lake.

iv) Using the anoxic version of the mass balance model, it was demonstrated that 100% of the potential anthropogenic TP supply must have entered Chub and Blue Chalk Lake if the observed concentrations of TP were to be explained.

v) Similarly, using the oxic version of the model it was demonstrated that 100% of the potential anthropogenic TP supply must have entered Dickie Lake. However, only 19% of the potential anthropogenic TP supply need enter Harp Lake to explain the observed TP concentration.

The reasons for the difference between Harp Lake and Chub, Blue Chalk and Dickie Lake are perhaps twofold, and relate to the nature of the surficial deposits around the lakes and the age of the cottages on the lakes. The fraction of the potential anthropogenic TP supply from lakeshore development that enters the lake can be expected to vary with the type of sewage system used, the degree to which the system is functioning properly and the available TP binding capacity of the soil between the sewage system and the lake. The latter relates to the type of soil or surficial deposit around the lake. While no detailed information specific to the ability of soils to bind or retain TP was collected in this study, Jeffries and Snyder (1983) reported on differences in surficial geology between the study lakes. The surficial geology of Chub and Dickie Lake is primarily composed of thin till and rock ridges (61% and 81% respectively) whereas minor till plains dominate in Blue Chalk and Harp (77% and 50% respectively). Relatively thick deposits of outwash (5%) and sand (4%) are found in the Blue Chalk and Harp basins respectively. The age of the cottages may be related to the proportion of the TP binding capacity of the soils that remains available, the cumulative anthropogenic TP from older cottages having saturated the available TP binding sites. The average age of cottages (Downing,

pers. comm.) around Harp Lake is less (1967) than it is around Chub (1961), Blue Chalk (1955) or Dickie Lake (1962). Additionally, it might be expected that the sewage disposal facilities (chiefly septic field systems) are in a better state of repair at Harp Lake because of their age.

C. EVALUATION OF THE EFFECTS OF LAKESHORE DEVELOPMENT ON AQUATIC BIOTA

Human activities associated with shoreline development of lakes may increase rates of nutrient supply to lakes. As the productivity and structure of aquatic communities may be nutrient dependent, increases in nutrient supply may have significant consequences for aquatic biota.

A variety of human activities may have detrimental impacts on aquatic life. Toxicants such as herbicides or petroleum residues may be introduced to lakes and impair the growth or survival of aquatic organisms. Littoral zone inhabitants may be sensitive to lake level manipulations or to siltation accompanying various activities in the watershed. Substantial impacts may accompany direct exploitation of particular communities such as harvesting of aquatic weeds or angling. Fishing may also indirectly influence benthic and plankton communities as removal of predators may change predation and grazing pressure on prey organisms.

The amount of development on the 15 study lakes was discussed in Section III.B.6. Human use ranged from 0-111 cap-yrs. $\text{yr}^{-1} \cdot \text{lake}^{-1}$. Expressed areally and volumetrically, respectively, human usage ranged from 0 to 1.52 cap yrs $\text{ha}^{-1} \text{ yr}^{-1}$ and from 0 to 0.24 cap yrs. $10^4 \text{ m}^{-3} \text{ yr}^{-1}$ (Table 33). Trophic status of lakes is usually strongly correlated with phosphorus supply. The fraction of the total phosphorus supply potentially attributable to cottage domestic waste systems was calculated for six of the lakes. It ranged from 0% in Jerry Lake to 55% in Harp Lake (Table 45).

In this section, impacts of development on communities of phytoplankton, macrophytes and zooplankton are examined. Impacts are assessed by examining correlations between parameters describing the state of the biological communities and indices of lakeshore development or by comparing the values of the parameters in sets of lakes of extremely different intensities of use.

1. PHYTOPLANKTON

Pearson's product moment correlation coefficients were calculated between indices of human usage of the study lakes expressed in total, volumetric and areal units and several characteristics of the ice-free period phytoplankton communities of the lakes (Table 65). Several of these characteristics are known to be highly correlated with lake water TP concentrations (Nicholls and Dillon 1978, Dillon et al. 1978, Nicholls et al. 1983) and are therefore considered to be excellent indicators of the trophic status of the lakes.

Of 18 correlation coefficients calculated, the only significant ($p < 0.05$) correlation was between chlorophyll *a* and volumetric use. However, only 28% of the variance in chlorophyll *a* data was explained by its relationship with volumetric use.

Table 65. Correlation coefficients (r) between six characteristics of the phytoplankton communities of the study lakes and measurements of human use. Phytoplankton data were input as averages of the ice-free seasons of 1976-1979.

Parameter	Use		
	Total	Areal	Volumetric
Total Biomass	0.36	0.32	0.44
Chlorophyll <i>a</i>	0.43	0.36	0.53
Maximum "Bloomer" Biomass ³	0.15	0.24	-0.04
No. of Chlorophyceae Genera	0.34	0.31	0.17
Species Richness ¹	0.39	0.33	0.24
Concentration of Dominance ²	-0.28	-0.30	-0.18

¹ calculated as $S-1/\ln N$; (Margaleff 1951), where S is the number of species (taxa) per sample and N the total number of individuals.

² calculated as $\sum (P_i)^2$ (Simpson 1949), where P_i is the proportion of the total sample belonging to the i^{th} species.

³ maximum biomass of bloom-forming blue-green algae.

Development effects were then examined (t-test) by comparison of the means of several phytoplankton community parameters of two groups of lakes, the three most highly developed lakes and the four completely undeveloped lakes (Table 66). There were no significant differences ($p > 0.1$) between the two groups for the six phytoplankton community parameters. Clearly, factors other than human use alone are responsible for shaping the phytoplankton communities of these lakes.

2. MACROPHYTES

To assess whether development affected macrophyte community composition or nutritional status, macrophyte communities were surveyed in eight of the study lakes that reflected the full range of human use. Standing stocks of macrophytes were determined in Red Chalk and Harp Lakes, an undeveloped and a heavily developed lake, respectively.

Table 66. Four-year means of several phytoplankton community characteristics in three lakes with the highest estimated human use (developed) and the four undeveloped lakes. The t value for the difference between the means of groups and probability value (p) for rejecting a hypothesis of equal means of groups are indicated.

Lakes	Total Phytoplankton Biomass (mm ³ L ⁻¹)	Total Chlorophyll (mg m ⁻³)	Maximum Blue-green "Bloomer" Biomass (mm ³ L ⁻¹)	Number of Chlorophyceae Genera	Species ¹ Richness	Concentration ¹ of Dominance
DEVELOPED						
Harp	0.66	2.75	0.94	32	80.6	0.09
Dickie	1.07	5.31	0.007	25	71.4	0.14
Walker	0.45	2.27	0.13	17	57.4	0.12
mean (s.d.)	0.73 (0.32)	3.4 (1.6)	0.36 (0.51)	24.7 (7.5)	69.8 (11.7)	0.12 (0.03)
UNDEVELOPED						
Jerry	0.79	3.09	0.10	25	65.4	0.20
Buck	0.52	2.27	0.005	18	54.4	0.14
Little Clear	0.45	2.15	0.23	20	56.9	0.15
Gullfeather	0.99	4.56	0.008	24	63.6	0.11
mean (s.d.)	0.69 (0.25)	3.0 (1.1)	0.08 (0.10)	21.8 (3.3)	60.1 (5.3)	0.15 (0.04)
t	0.18	0.41	1.08	0.71	1.51	1.32
p	0.80	0.60	0.20	0.50	0.10	0.20

¹ Defined in Table 48.

Table 67. Species richness, tissue concentrations in *E. septangulare* and macrophyte community biomass (with 95% confidence interval) in lakes. TP is the mean of 4 annual means (June 1976 - May 1980). Annual TP concentration and areal use of lakes as indicated.

Lake	Areal Use (M-yr ha ⁻¹ yr ⁻¹)	TP (mg m ⁻³)	Richness (number of species lake ⁻¹)	Mean Tissue Concentration ¹ (mg (g d.w.) ⁻¹)				Total Biomass (g m ⁻²)
				TP		TN		
				Plant	Sediment	Plant	Sediment	
Chub	0.13	12.2	18	0.64	0.32	18.5	0.73	-
Crosson	0.085	11.7	19	1.00	0.29	19	1.10	-
Dickie	1.19	12.8	19	0.73	0.27	16	1.70	-
Gullfeather	0	13.4	19	0.74	0.30	17	0.55	-
Harp	1.52	7.8	18	0.47	0.35	14.2	0.67	73.6 ± 16.0
Little Clear	0	11.2	16	0.59	0.25	19	2.10	-
Red Chalk	0.033	5.7	13	0.69	0.56	14.3	0.51	59.2 ± 7.2
Solitaire	0.13	6.4	6	0.67	0.12	20	0.50	-

¹ Correlation coefficients between plant TN and sediment TN, plant TP and sediment TP, plant TP and water TP are 0.22 ($p < 0.10$), 0.06 ($p < 0.10$) and 0.35 ($p < 0.10$), respectively.

Divers recorded 33 taxa of submersed and floating-leaved vascular macrophytes, macroalgae (charophytes) and bryophytes in the eight lakes. All communities were dominated by "isoetid" or rosette species, particularly by *Eriocaulon septangulare* and *Lobelia dortmanna*, as is typical of softwater, oligotrophic lakes (Moeller 1975, 1978, Moyle 1945). Species richness was not correlated with areal human use (Table 67).

Vegetation cover was typically patchy and densest stands were largely confined to areas adjacent to inflowing streams. A distinct vertical zonation of species occurred in all lakes irrespective of development. Isoetids and floating-leaved species (*Nuphar*, *Nymphaea*, *Brasenia*) colonized water 3m deep. Bryophytes and *Nitella* sp. colonized deeper water.

Gerloff and Krombhotz (1966) have indicated that the growth of vascular macrophytes will be limited by phosphorus and nitrogen when tissue concentrations are $<1.3 \text{ mg P g}^{-1}$ and $<13 \text{ mg N g}^{-1}$, respectively. Tissue concentrations of N and P in *E. septangulare*, the dominant species in all lakes are indicated in Table 67. Plant levels were not correlated ($p > 0.05$) with water or sediment nutrient levels and did not increase in developed lakes. Assuming applicability of the Gerloff and Krombhotz criteria, *E. septangulare* was P-limited in all lakes regardless of development. Tissue levels of N indicated that supplies of this nutrient were sufficient.

Given that development had not increased nutrient availability, it is perhaps not surprising that total macrophyte community biomass was similar in Harp Lake, a heavily developed lake ($73.6 \pm 16 \text{ g m}^{-2}$) and in the relatively undeveloped Red Chalk Lake ($59.2 \pm 7.2 \text{ g m}^{-2}$, Table 67).

These comparisons indicate indirect effects of shoreline development operating through changes in nutrient

supply have no detectable significant effects on the composition, standing stocks or nutritional state of aquatic macrophyte communities of the lakes. Other direct interventions related to development, such as beach clearing, would, of course, have negative impacts on vascular macrophytes.

3. ZOOPLANKTON

Averages for 1977 to 1979 for ten zooplankton community parameters were assembled for the 15 study lakes (Table 68). To test whether development influenced zooplankton community structure or standing stocks, t tests of difference between means were performed on two groups of lakes, the three most highly developed lakes (Harp, Dickie and Walker) vs. the four completely undeveloped lakes (Jerry, Buck, Little Clear and Gullfeather).

Development might have been expected to increase nutrient supply, yet neither total nor herbivorous zooplankton biomass, nor the biomass of Calanoida, Cyclopoida or Cladocera were significantly higher ($p > 0.05$) in the developed lakes. Development similarly did not influence the average richness of the zooplankton communities. Richness averaged 9-10 species per collection in each group of lakes.

The proportional biomass of herbivores vs. carnivores or of cladocerans and copepods were not different in the two lake groups. Finally, the zooplankton were of similar average size in the developed and undeveloped lakes, indicating that changes in zooplankton predators or prey produced by development, did not have a large impact on zooplankton size structure.

The range in zooplankton parameters was generally greater among the undeveloped than the developed lakes (Table 68). This variability is clearly produced by some factor other than human usage of the lakes.

Table 68. Comparison of various zooplankton parameter means between three developed study lakes (Harp, Dickie, Walker) and 4 undeveloped lakes (Jerry, Buck, Little Clear, Gullfeather). t value and probability value for rejecting the hypothesis of equal means are indicated.

Lake Group	Areal Use (m-yr ha ⁻¹ yr ⁻¹)	Crustacean Zooplankton Biomass (mg m ⁻³)					Percent Biomass			Richness (No. of species/ collection)	Size (µg/ animal)
		Total	Herbi- vore	Cala- noida	Cyclo- poida	Clado- cera	Herbi- vore	Clado- cera	Cope- poda		
Developed mean	1.21	47.4	37.6	12.6	9.4	23.9	79	50	50	9.8	0.73
lowest value	0.93	40.2	30.1	11.8	8.0	19.8	75	49	47	9.1	0.66
highest value	1.5	53.5	43.2	13.5	11.8	25.7	81	53	51	11.0	0.85
Undeveloped mean	0	61.6	45.9	5.2	14.8	40.3	74	61	39	9.1	0.66
lowest value		32.5	25.8	1.5	7.0	16.8	62	35	24	8.2	0.54
highest value		95.1	74.1	13.7	19.9	72.7	79	76	65	10.4	0.71
t	8.49	0.86	0.6	2.14	1.59	1.04	1.05	1.01	1.01	0.93	1.11
p	0.001	0.50	0.60	0.10	0.20	0.40	0.40	0.40	0.40	0.40	0.40

IV. DEVELOPMENT AND CALIBRATION OF MODELS FOR PREDICTING THE EFFECTS OF DEVELOPMENT ON LAKE TROPHIC STATUS

In Part III of this report, a combination of existing and newly formulated models were used to evaluate the contribution of lakeshore development to the TP content of the "A" lakes. In this section, generalized models are developed and calibrated for predicting the effects of shoreline development on the trophic status of soft-water lakes in the Precambrian regions of Ontario.

The variables that have traditionally been used to describe the trophic status of lakes include the chlorophyll *a* concentration (a measure of phytoplankton abundance), phytoplankton biomass, phytoplankton species composition (especially the presence or absence of indicator species, e.g., bloom-forming blue-green algae), water clarity (usually measured as the Secchi disc depth), and areal hypolimnetic oxygen depletion rate (AHOD). These variables have been focused on for several reasons: a) they are "state variables"; that is, they integrate many components of the lake system, b) they are relatively easily measured, and c) they represent measures of trophic status useful to the layman since they are readily visible or have readily visible effects (e.g., high AHOD may result in fish kill).

The conceptual model that linked these state variables to the water and TP mass balances of lakes was outlined in Figure 1. In the following sections, models suitable for predicting water and TP balances are formulated, and linkages between these models and state variables defining trophic status are presented.

The possibility that acidic deposition has altered the relationships linking trophic status parameters is also considered in this section. The chlorophyll *a*, phytoplankton biomass and water clarity models are derived with consideration of both nutrient enrichment and acidification parameters, as is the zooplankton biomass model discussed in Section V.

A. HYDROLOGIC MODELS

All important components of the hydrologic regimes of the "A" lakes were directly measured in this study as this information was needed to determine the natural inputs of total phosphorus to the lakes. In fact, one criterion for selecting these study lakes was that collection of such data was technically feasible. It is costly, time consuming and may not always be technically feasible to assemble measured hydrological data for all lakes on which development may occur. It would therefore be of great aid to lake managers if models which accurately predict important components of the hydrologic cycle of Precambrian Shield

lakes could be developed. The possibility of constructing such models is investigated in this section.

Groundwater flux and changes in lake storage were quantitatively unimportant for the six "A" lakes (Section III.B.2) as previously observed for other Precambrian lakes (Schindler et al. 1976, Newbury and Beaty 1977, Scheider 1983). Precipitation to the lake surface and terrestrial runoff were the major sources of water to the lakes. Loss via the lake outlet and evaporation were the major losses. If these four terms of the water balance equation for lakes could be accurately predicted, then the hydrologic regime of most lakes would be well characterized.

For a terrestrial watershed, precipitation, runoff and evapotranspiration are the three important components in the annual water balance. As the time interval of interest decreases (years to months to weeks), changes in storage (either in lakes or in soil) become more important in the water balance. Given the primary interest in using the data to predict average water quality changes, annual timesteps are generally adequate. For annual timesteps, changes in storage may be ignored.

1. MODELLING PRECIPITATION DEPTH

The simplest method of predicting precipitation depth at a given location is to use the measured value for that time at a nearby meteorological station. Many such stations operate continuously in Ontario (Hydrological Atlas of Canada 1978). Precipitation depth was relatively uniform between the seven stations in the study area, (i.e., no significant differences in depth among stations on an annual basis or on a monthly basis in 10 of 12 months), indicating that annual and monthly precipitation depth at a given lake in the study area may be estimated using data from the closest meteorological station. To improve estimates for shorter time durations (daily) or across areas for which precipitation depth does vary, a weighted average of the two closest meteorological stations (Viessman et al. 1977) may be calculated as follows:

$$P_{i1} = \sum_j P_{ij} \cdot (W_j / \sum_j W_j) \quad (34)$$

where: P_{i1} = precipitation for day *i* derived for a given location number 1
 P_{ij} = precipitation for day *i* at location *j*
 $W_j = 1/D_j^2$
 D_j = distance from given location to meteorological station *j*.

Other methods of obtaining precipitation depth using data from several surrounding stations are the Thiessen polygon method and the isohyetal method (outlined in Gray 1970).

No reliable method exists for predicting precipitation in the future; however, a range of possible values at a particular location can be generated using long-term records such as those summarized in the Hydrological Atlas of Canada (1978).

2. MODELLING RUNOFF AND EVAPOTRANSPIRATION

Runoff from the terrestrial basin supplied the largest quantity of water to the study lakes. Loss via the outlet was the major loss (Tables A1-A6). There are many approaches to predicting surface runoff (summarized in McMahon 1979). The strengths and weaknesses of five techniques are summarized below, along with their required data inputs.

a) Annual runoff may be assumed to equal long-term average runoff. Coulsen (1967), Pentland (1968) and Hydrological Atlas of Canada (1978) present isolines of average runoff drawn from information at many gauged rivers and streams in Ontario. Errors in estimates generated using this method can arise due to local variability in precipitation depth, evapotranspiration rates and catchment characteristics such as physiography, forest cover and soil type. Of course, as the data are long-term average values, errors in prediction will occur in any given year because of climate variability. Pentland (1968) also gives long-term mean runoff isolines for Ontario on a monthly basis. However, one can expect progressively greater divergence between predicted and actual runoff the shorter the time interval for prediction. Year-to-year variability in monthly runoff in the 1976-1980 period was substantial for the "A" lakes (Table 69), particularly in October and March.

b) A second method for predicting annual runoff from ungauged watersheds in Muskoka-Haliburton from 1976-80 is to use the mean runoff of the gauged "A" lake and export watersheds measured in the same year. This overcomes problems associated with the use of long-term mean runoff data. The fractional distribution of annual runoff between months differed in the headwater watersheds compared to the lake outlets (Table 69). Predictions for ungauged watersheds should therefore employ appropriate proportions of annual runoff.

c) The second method corrects predictions of runoff only for year-to-year differences in some meteorological characteristics, but does not account for differences among watersheds. Multiple linear regression was used as a technique to attribute the variability in streamflow characteristics among watersheds to various watershed characteristics. As expected, total annual discharge was highly related to watershed area in each of the four years ($r^2 \approx 0.98$). The slope of the regression line between watershed area and annual discharge is in effect the mean runoff, so predicting annual streamflow in this manner yields similar (differing by the value of the intercept) results to those predicted by the second method.

Four watershed characteristics explained 45% of the variance in mean (1976-1980) annual runoff ($r^2 = 0.448$; $n = 36$). Mean annual runoff (R , mm yr^{-1}) was a function of stream length (L , m), % of the watershed area underlain by organic deposits (OR , %), % of the watershed area underlain by shallow till (ST , %), and stream length/watershed area (L/A , m^{-1}) as follows:

$$R = 355 + 0.043(L) + 6.29(OR) - 0.91(ST) + 1.98(L/A) 10^4$$

The above three techniques use measured runoff values at gauged watersheds to estimate runoff at ungauged watersheds. The estimates use either long-term mean values or apply only to the 1976-1980 period. More general models are necessary to allow prediction of runoff for any given set of climatic conditions or to predict runoff for watersheds with different characteristics than those of the "A" lakes.

d) If change in storage and groundwater flux are assumed to be zero (a reasonable assumption on an annual timescale), then runoff will equal the difference between precipitation and actual evapotranspiration. Hence annual runoff may be predicted from measured annual precipitation if independent predictions of evapotranspiration are available.

Evapotranspiration may be estimated in four ways. The mass balance method keeps an account of the components of the water balance equation and arrives at evapotranspiration by difference. The energy balance method requires measurement of components of an

Table 69. Mean % annual stream discharge occurring in each month for 23 headwater streams and 6 lake outlets for the 1976-1980 period.

Month	23 Headwater Streams			
	1976-77	1977-78	1978-79	1979-80
June	1.65	0.77	2.18	2.40
July	3.19	0.58	0.63	0.94
Aug.	0.95	0.77	2.26	1.47
Sept.	0.80	2.72	2.58	1.19
Oct.	1.71	10.03	6.48	8.33
Nov.	4.71	14.51	8.91	16.39
Dec.	5.02	9.20	6.36	10.18
Jan.	2.17	3.95	3.50	7.31
Feb.	1.50	1.75	2.13	1.58
Mar.	39.82	2.95	26.56	10.20
Apr.	34.22	39.75	28.57	34.81
May	4.23	12.91	9.82	5.14

Month	6 Lake Outlets			
	1976-77	1977-78	1978-79	1979-80
June	2.51	0.47	3.20	3.61
July	1.97	0.34	0.50	0.39
Aug.	0.69	0.46	1.33	1.05
Sept.	0.31	0.71	2.15	1.40
Oct.	0.47	8.34	4.41	6.19
Nov.	2.74	15.09	8.06	15.47
Dec.	6.97	12.85	8.71	13.82
Jan.	3.72	7.20	7.11	9.57
Feb.	2.89	3.46	3.61	3.44
Mar.	29.06	3.88	21.65	8.05
Apr.	43.47	29.69	27.08	31.37
May	5.11	17.48	12.18	5.67

energy balance equation to calculate evapotranspiration. The vapour transfer method considers the physical process of evaporation, i.e., evaporation is proportional to the vapour pressure gradient at the surface. Other indirect methods make use of meteorological information available at most climatic stations. The first three methods require many forms of instrumentation to obtain values of necessary input parameters. As this instrumentation is generally not in place, these methods are unsatisfactory for predicting evapotranspiration for most watersheds.

Several indirect methods are available. Penman (1948) developed a semi-empirical formula to predict free water evaporation which he multiplied by a constant to obtain actual evapotranspiration. Turc (1954, 1955) developed formulae to relate annual actual evapotranspiration to precipitation. Application of Turc's formula for shorter time periods (<annual) requires a crop factor. Finally, Morton (1965, 1968, 1969, 1971, 1976, 1978) and Morton et al. (1980) developed an approach to predict actual evapotranspiration from information on air temperature, dew point temperature and the ratio of observed to maximum possible sunshine duration or the observed global radiation. The simplicity and availability of the data required to run the model and the good agreement with long-term water balance estimates of areal evapotranspiration (Morton 1978) made this the most attractive model.

Using Morton's method, estimates of evapotranspiration were calculated for the Harp and Jerry Lake watersheds. These estimates were compared with direct measurements (a weighted average water loss over all watersheds) derived from the measured water balance. Discrepancies were observed on a monthly basis (Figure 13) but this was expected because changes in storage are not considered by the water balance estimate. However, over the 1976-1980 period, the total difference in evapotranspiration estimates was only 31 mm (1957 mm estimated by the water balance method vs 1988 mm estimated by Morton) or <2% (Figure 14). The good agreement between the two methods gives confidence in the use of Morton's method to estimate evapotranspiration and hence runoff on an annual basis. However, for shorter time steps (e.g., monthly), changes in storage must also be taken into account if runoff is to be predicted from precipitation and evapotranspiration.

Because the input data to Morton's model were the same for all Harp and Jerry Lake watersheds, the model predicted that all watersheds would have the same evapotranspiration and, because the precipitation depth is also the same at the two lakes, the same runoff. The observed range in mean (1976-1980) annual runoff (0.29-0.56 m yr⁻¹) among the watersheds of Harp and Jerry Lake, presumably due to differences between watershed characteristics, cannot be addressed using Morton's model.

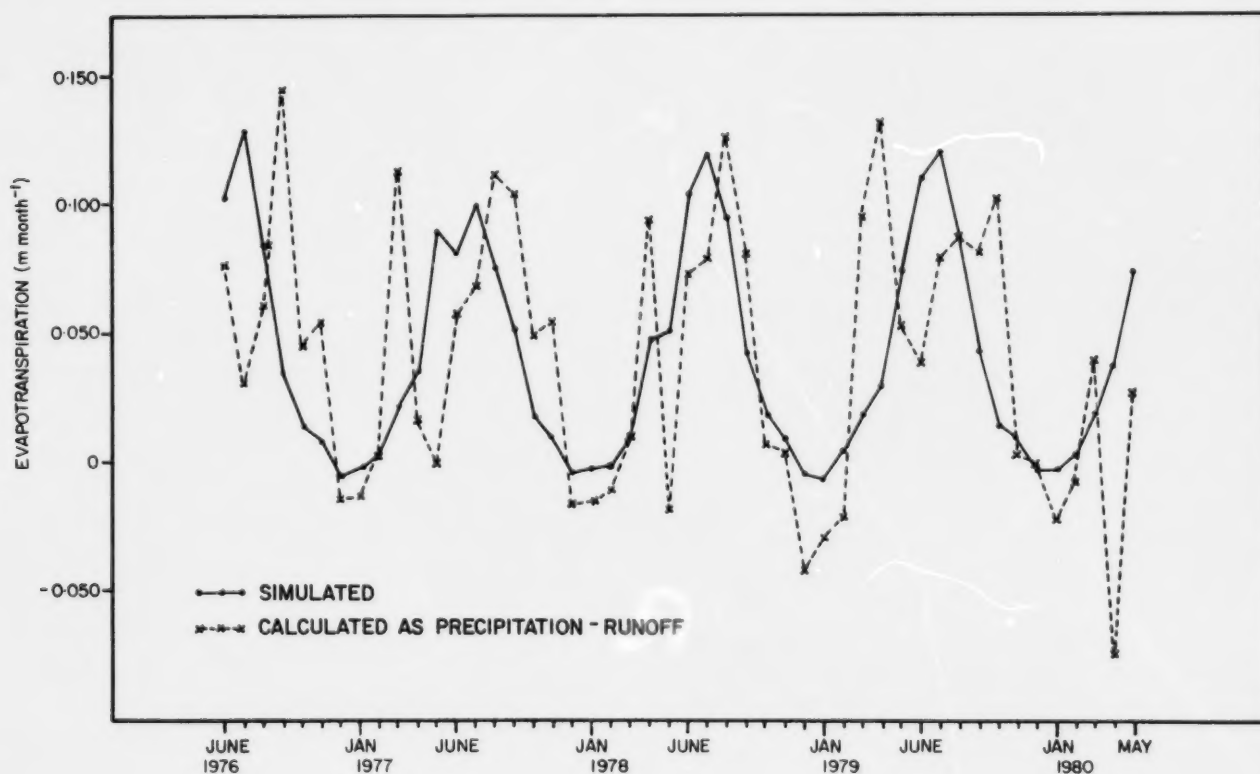


Figure 13 Monthly evapotranspiration losses estimated using the method of Morton et al. (1980) (solid line) compared to losses calculated as the difference between observed precipitation and runoff (dashed line) for watersheds of Harp and Jerry Lake (1976-1980).

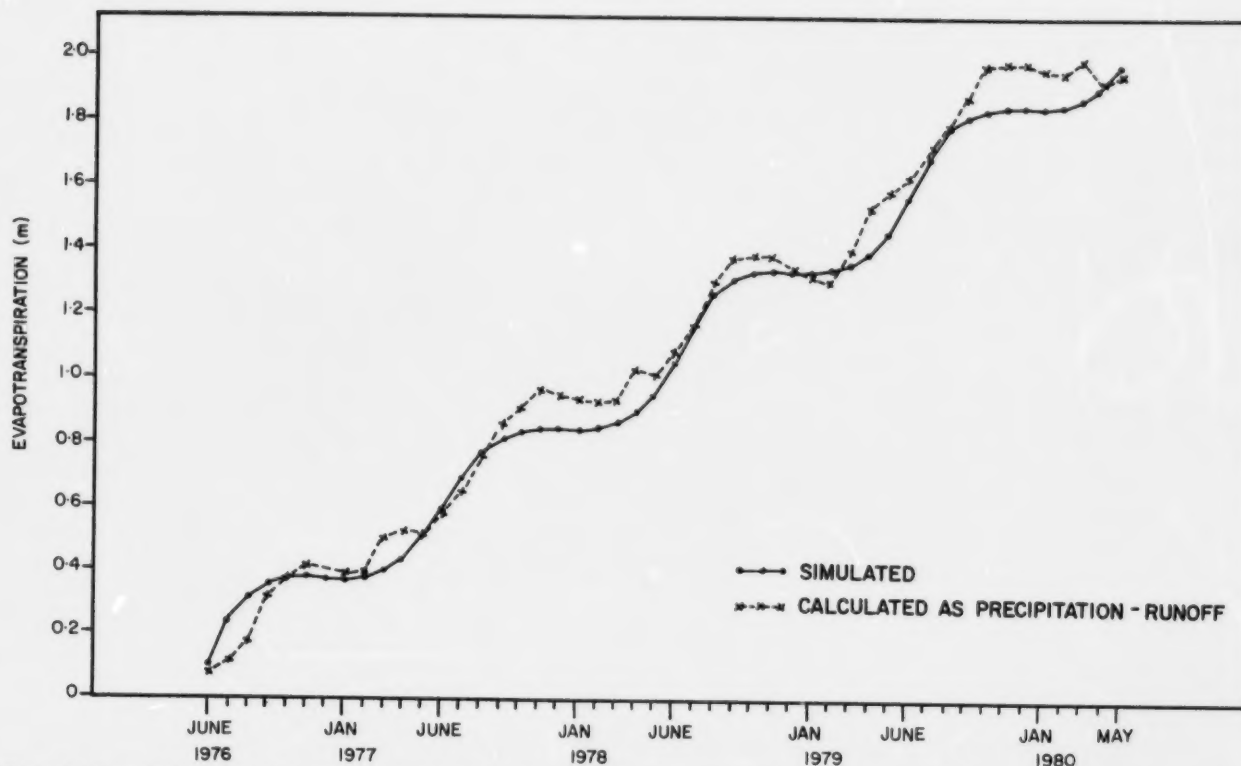


Figure 14 Cumulative plot (1976-1980) of monthly evapotranspiration losses estimated after Morton et al. (1980) (solid line) and losses estimated as the difference between observed precipitation and runoff (dashed line) for watersheds of Harp and Jerry Lake.

e) Finally, runoff is the output of a range of process models (many are summarized in Fleming 1975) which simulate the movement of water through the drainage basin. These models predict flow on a very fine time scale (daily) for any given set of climatic conditions. Several such models have been used in chemical mass balance studies (e.g., Christopherson et al. 1982, Chen et al. 1982). A model developed by the Tennessee Valley Authority (1972) was adapted for use in Precambrian areas of Ontario (Goebel and Thysen 1982, Goebel 1982, 1983). This model was chosen because it provides daily flow estimates using readily obtainable meteorological data as inputs (Logan 1980).

The Tennessee Valley Authority (TVA) model is based on the concept of water balance (Figure 15). All incoming moisture is allocated by the model such that precipitation - evapotranspiration - streamflow - Δ storage = 0. The model operates in a cascading fashion to allocate precipitation to the various compartments shown in Figure 15. The ultimate fate of incoming precipitation is loss to evapotranspiration or streamflow. Deep seepage losses are set to zero for the study watersheds.

The model considers precipitation to be rain if the mean daily air temperature $\geq 1.5^\circ\text{C}$. Rain and snow are first lost to interception then cascade to the next compartment. The quantity of accumulated snow which melts in a day is a function of mean daily air temperature and rainfall as follows:

$$\text{melt (mm)} = 1.3 (\text{air temp. } ^\circ\text{C}) (\text{if air temp. } > 1.5^\circ\text{C}) \quad (35)$$

$$\text{melt (mm)} = (3.5 + 0.012 \times \text{precip.}) \text{ air temp. } ^\circ\text{C} + 1.2 \quad (\text{if air temp. } > 1.5^\circ\text{C, if rain}) \quad (36)$$

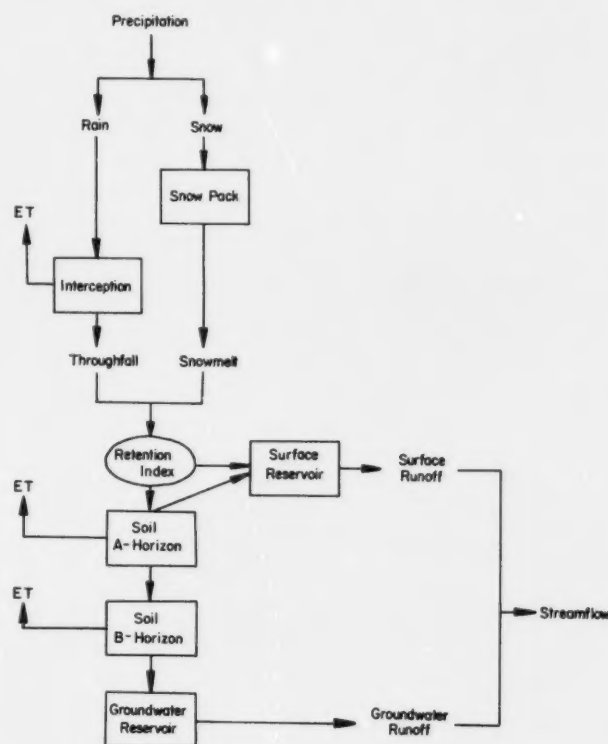


Figure 15 Schematic outline of the modified TVA streamflow model.

The snowpack itself is assumed to have a 3% water retention capacity. This part of the model shifts the time when precipitation input is available for runoff routing.

In either case, (rain or snow), the incoming precipitation, after losses to interception and storage or release by the snowpack, is potentially available for surface runoff. The amount allocated to surface runoff is between 0 and 100%, depending upon the retention index. If the soil is saturated with moisture, the retention index assumes a value of 0 and all incoming precipitation is available for runoff. Conversely, under dry soil conditions the value of the retention index approaches 1 and little water is available for surface runoff.

The remainder of the water is potentially available for groundwater runoff (baseflow). It first cascades into the A soil compartment (maximum storage capacity = 25 mm). If this compartment is saturated, the remaining water is routed to surface runoff. Water from the A soil compartment is transferred to the B soil compartment and ultimately into the groundwater reservoir. Losses from the groundwater reservoir are calculated from a groundwater recession curve. Groundwater plus surface runoff combine to yield total streamflow. The model contains several routing equations which determine the rate of surface and groundwater runoff.

In addition to streamflow, water is lost by evapotranspiration (Morton et al. 1980). Evapotranspiration demands are first met by the water stored in the interception compartment, then by water from the A soil compartment and finally by water from the B soil compartment.

The data required to utilize the model are daily precipitation, mean daily temperature and the data needed to compute evapotranspiration after Morton et al. (1980). In addition, the model requires that values for a number of parameters be fixed. These include such parameters as the beginning of the spring, summer, fall and winter seasons, maximum moisture storage by the interception, A and B soil compartments etc. There are nine parameters which are optimized in the model. An optimization routine which sets the value of these nine parameters by minimizing the sum of squared differences between the observed and predicted streamflow is included in the model calibration procedure. The model runs on a time period of October – September. Prior to this, there is a 3-month lead in period to assist in setting values of various model parameters.

The modified TVA model was calibrated using data for the period Oct. 1, 1976 to Sept. 30, 1979 for the 21 gauged inlets to the "A" lakes. Overall correlation coefficients for the 3 year period ranged from 0.57 at Harp inlet 3 to 0.76 at Chub inlet 1. The predicted and observed daily runoff for Harp inlet 4 are shown in Figure 16 (a,b,c) as a sample result. Major fall storm events were correctly simulated by the model as were low runoff values during the winter and summer periods. Runoff during the spring period was modelled quite well for 1977 and 1979 but the simulated runoff did not agree with the observed runoff in the spring of 1978. The reason may be that in 1978 the spring runoff occurred relatively late. In the TVA model, values of various parameters vary seasonally but the

beginning of the seasons is fixed. A varying season parameter would probably improve the model results.

The model was tested in two ways: by extending the runoff records in time for watersheds on which the model had been calibrated and by predicting runoff at watersheds for which the model had not been calibrated. In the former test, the model was used to predict runoff for the period Oct. 1, 1979 to May 31, 1980 for the 21 gauged inlets of the "A" lakes. The predicted values of runoff were compared to observed data. Values of the correlation coefficient between predicted and observed runoff ranged from 0.55 to 0.89. A sample result (Harp inlet 4) is shown in Figure 16 (d).

In the second test, there were no model calibration runs against observed runoff data to set values for the nine model parameters and these had to be estimated by other means. A multiple linear regression analysis using 21 estimates of each of the nine model parameters (data obtained from model calibration runs on the 21 gauged inlets to the "A" lakes) was carried out using measures of the geological and physical characteristics of the water-

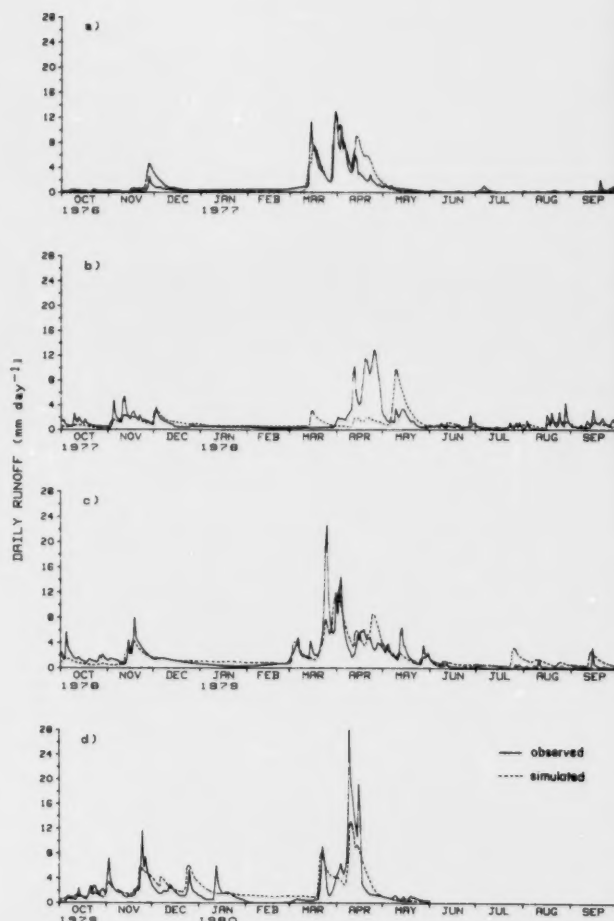


Figure 16 Comparison of simulated (TVA model, dashed line) and observed (solid line) daily runoff (mm day^{-1}) of Harp inlet 4 for the period during which the model was calibrated (Oct. 1976 – Sept. 1979) and for a test period (Oct. 1979 – May 1980).

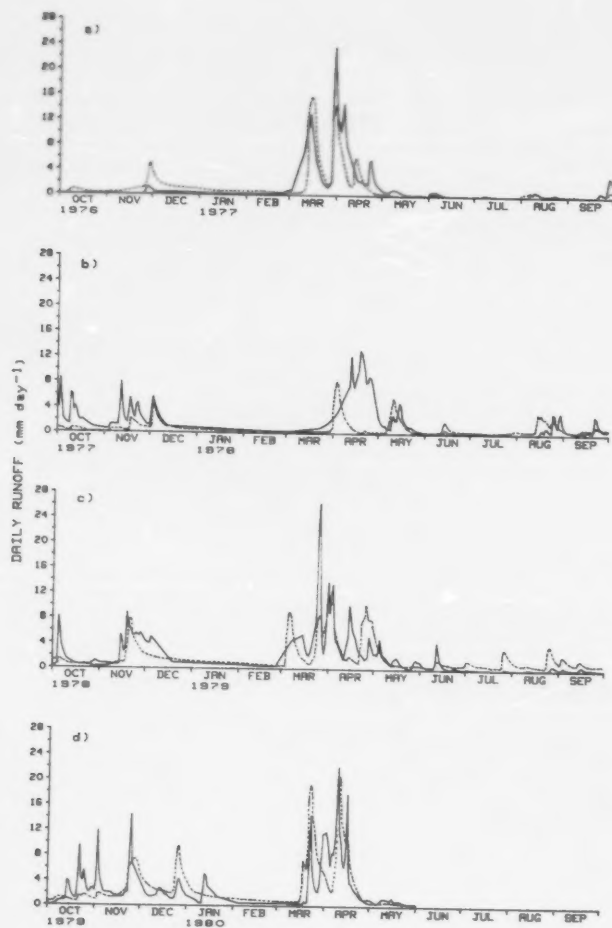


Figure 17 Comparison of simulated (TVA model, dashed line) and observed (solid line) daily runoff (mm day^{-1}) for Paint inlet 1 for the test period Oct. 1976 – May 1980.

sheds as independent variables. Although the results of all the regression analyses are not reported here, it was found that four independent variables (watershed gradient, and the % watershed area covered by woodland, organic deposits and exposed bedrock) explained 42-92% of the variance in some model parameters. The mean value of the parameter was used where no relationship with the independent variables was found. Using these relationships, the TVA model was tested against observed runoff data for eight of the nine "export" watersheds for the period Oct. 1, 1976 – May 31, 1980. Values of the correlation coefficient between predicted and observed runoff ranged from 0.05 to 0.91. As was observed in the calibration runs on the 21 gauged "A" lake inlets, the poorest agreement between the predicted and observed runoff generally occurred in the spring of 1978. Results for Paint inlet 1 are shown in Figure 17 as a sample.

3. MODELLING EVAPORATION

As with precipitation and runoff, lake evaporation may be predicted using long-term mean annual records (Hydrological Atlas of Canada 1978). Of course, year-to-year variations are not addressed by this method. The

evaporation isopleths are based on modified pan evaporation data, and are available from certain climatic stations on an annual as well as mean long-term annual basis.

Lake evaporation during 1976-80 for lakes in Muskoka-Haliburton may also be predicted using the mean annual evaporation measured for the "A" lakes (Table 12). This technique is acceptable for prediction on an annual basis because no significant ($p > 0.05$) differences occurred in lake evaporation between the seven lakes. However, it is less justifiable on a monthly basis because significant ($p < 0.05$) differences in evaporation between lakes occurred in some months. This method of predicting evaporation is less certain on a broader spatial scale.

A more general method of predicting lake evaporation based on climatological data was presented by Morton (1979) and Morton et al. (1980). The data needed are the same as those for his evapotranspiration model. Using Harp Lake as an example, evaporation estimated by the Morton method and by the energy balance method showed similar seasonal patterns (Figure 18), although Morton's method estimated higher evaporation rates during the early part of the open water period (April – July) and lower evaporation rates during the later period (August – December). The annual evaporation estimates agreed closely, Morton's method being 2.5%-7.6% higher, in part because the energy balance method assumed zero evaporation during the period of ice cover.

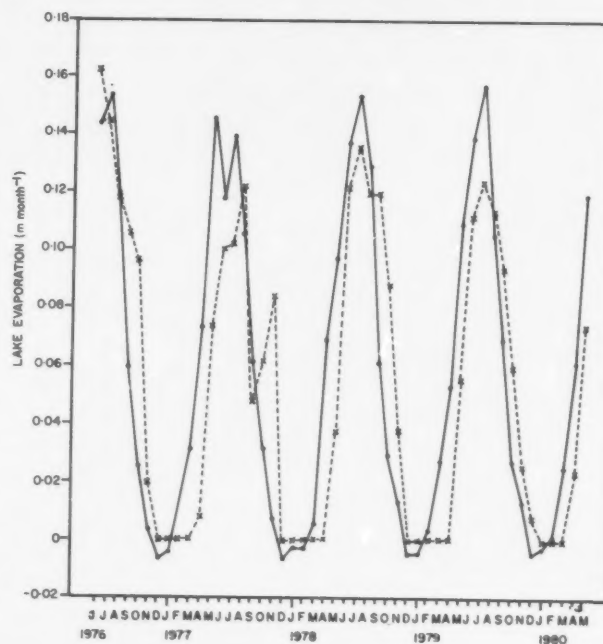


Figure 18 Comparison of monthly lake evaporation estimated by the energy balance method (dashed line) and by the method of Morton et al. (1980) (solid line) for Harp Lake (1976-1980).

B. MODELLING THE EXPORT OF TOTAL PHOSPHORUS

The flux of total phosphorus was measured from all major watersheds of the "A" study lakes. However, because of time, cost or technical limitations it may not always be feasible to obtain these measurements for other lakes.

Therefore, models to predict TP export from readily obtainable information would be of assistance to lake managers. Modelling the TP export from watersheds is discussed in this section with specific reference to developing models for application to the "B" lakes. Analysis of the TP export data for the 21 gauged "A" lake watersheds and nine "export" watersheds (Section III.B.5) demonstrated that spatial variability in TP export was greater than year-to-year variability. Emphasis for modelling the TP export was placed accordingly.

Earlier attempts to relate TP export to watershed characteristics (Vollenweider 1968, Dillon and Kirchner 1975a, Kirchner 1975) have been discussed previously. The new models attempted to refine these relationships, primarily for watersheds classified by Dillon and Kirchner as "forested, igneous", by considering an expanded set of watershed characteristics. The stepwise strategy for multiple linear regression was used to derive the models (Draper and Smith 1981). Forty-six variables representing hydrologic, geologic and land use characteristics were employed as independent variables and TP export from 21 "A" lake watersheds and nine additional "export" watersheds as dependent variables. Data on land use and geology were assumed constant over the four-year period (1976-1980). The annual TP export data were summarized in Tables 22 and 23. Table 70 lists the 46 variables describing watershed characteristics.

Table 70. Variables used in regression analysis on annual export of total phosphorus.

Type of Variable	Source of Data	Variable (V) Number	Definition of Variable
Hydrology	*	1.	Annual precipitation (mm)
	*	2.	Annual runoff (mm)
	++	3.	Yield (ratio) $[= V2/V1]$
	*	4.	Annual maximum mean daily discharge/mean annual mean daily discharge (ratio)
	*	5.	Annual base flow/total annual flow (ratio)
	*	6.	Spring flow [March + April + May total/total annual flow] (ratio)
Geology	**	7.	Area of watershed (m ²)
	+	8.	Grade (% slope)
	+	9.	Runoff coefficient
	**	10.	Length of gravel roads (km)
	+	11.	Stream length of main channel (m)
	++	12.	Area/stream length (m ² /m)
Bedrock geology	**	13.	Biotite gneiss (% total area)
	**	14.	Amphibolite and schist (% total area)
	**	15.	Migmatite (% total area)
	**	16.	Marble (% total area)
	(a) **	17.	Meta-arkose and minor marble (% total area)
	**	18.	Biotite gneiss and arkose marble (% total area)
	(a) **	19.	Diorite (% total area)
	++	20.	Carbonate bedrock (% total area) $[V16 + V17 + V18]$

Type of Variable	Source of Data	Variable (V) Number	Definition of Variable
	++	21.	Silicate bedrock (% total area) $[100 - V20]$
	++	22.	Mafic bedrock (% total area) $[V14]$
	++	23.	Felsic bedrock (% total area) $[V13 + V15 + V19]$
Surficial geology	**	24.	Carbonate till (% area)
	**	25.	Minor till plain (% area)
	**	26.	Thin till and rock ridges (% area)
	**	27.	Peat (% area)
	**	28.	Bedrock (% area)
	**	29.	Outwash plain (% area)
	**	30.	Sand (% area)
	(a) **	31.	Esker (% area)
	(a) **	32.	Drumlin (% area)
	**	33.	Pond (% area)
	++	34.	Deep overburden (% area) $[V25 + V29 + V30 + V31 + V32]$
	++	35.	Shallow overburden (% area) $[V26 + V28]$
	++	36.	Organic overburden (% area) $[V27 + V33]$
Relative abundance of nutrient	***	37.	Sources (0 = none, 1 = some, 2 = many)
	***	38.	Sinks (0 = none, 1 = some, 2 = many)
sources and sinks	***	39.	Absence of sources or sinks (0 = presence, 1 = absence)
Land use	+	40.	Flat woodland (% area)
	+	41.	Rolling woodland (% area)
	+	42.	Hilly woodland (% area)
	+	43.	Pasture (% area)
	(a) +	44.	Cultivated (% area)
	+	45.	Water (% area)
	+	46.	Bog (% area)
	+	47.	Wetland (% area)
	(a) +	48.	Urban (% area)
	(a) +	49.	Flat bedrock (% area)
	+	50.	Rolling bedrock (% area)
	+	51.	Hilly bedrock (% area)
	+	52.	Total bedrock (% area)

Sources of Data

- * Ministry of the Environment: data obtained from periodic measurements and computed on an annual basis.
- ** Ministry of the Environment: data obtained from air photo interpretation, on-site validations, and collaboration with published geology maps.
- *** Ministry of the Environment: data obtained through inspection of air photos.
- + Ministry of Transportation and Communications: data obtained by airphoto interpretation.
- ++ Data computed from other variables.
- (a) Variable was eventually deleted from the analysis as it represented a zero vector or near-zero vector in the regression analysis.

Two tiers of models were constructed, one using all independent variables and a second excluding the hydrologic variables. Each tier of models included a regression model for each of four years (1976-77, 1977-78, 1978-79,

1979-80) as well as a model for the entire four-year period. No analysis of residuals was done. Missing data were deleted from the analysis on a listwise basis only.

Using TP export, hydrologic, geologic and land use data from all four years (1976-1980), the stepwise procedure (at the 90% confidence level) produced a final equation after five steps which accounted for 51% ($r^2 = 0.51$) of the variance in TP export. Annual TP export ($\text{mg m}^{-2} \text{yr}^{-1}$) was predicted as a function of peat (% area), runoff (mm yr^{-1}), grade (% slope), bedrock (% area), and pasture (% area). These results are given in Table 71, model A.

Using TP export, geological and land use data from all four years, a model was constructed (at the 90% confidence level) after four steps which accounted for 46% of the variance in TP export ($r^2 = 0.456$). As might be expected, the variables entered the equation in the same order as the previous model. The estimate of annual TP export ($\text{mg m}^{-2} \text{yr}^{-1}$) was predicted as a function of peat (% area), grade (% slope), bedrock (% area), and outwash plain (% area). The results are given in Table 71, model B.

Although the results of the regression analyses for data from individual years are not reported here in detail, it should be noted that these models were simpler (i.e., fewer predictor variables entering the regression equations) and less powerful (r^2 values < 0.5) than the models constructed on all four years' data. The variables peat and organic overburden were important predictors of annual TP export in all the models, both the single-year models and the four-year composite models. They usually entered first in the stepwise regression procedure and in some cases were the only significant variables in the single-year models.

C. MODELLING THE TP CONCENTRATION IN LAKES

The trophic status of most Precambrian Shield lakes is governed by the concentration of TP in the lake, which in turn is a function of the lake's TP mass balance, hydrologic budget and morphometry (Figure 1). Models which link these parameters to predict changes in the TP concentration of a lake have been discussed in Section III.B.10 and only a brief review will be given here.

Early mass balance models to predict the chemical content of lakes were reviewed by Dillon (1974). A simple model to predict the TP concentration in a lake at steady-state was expressed (equation 16) in the following

form:

$$[\text{TP}] = J_T / V\phi$$

Alternatively, if sedimentation loss of TP is expressed in terms of a sedimentation velocity (v), the model was expressed (equation 18) as:

$$[\text{TP}] = J_T / (Q + vA)$$

A third model formulation (equation 19) employed a retention coefficient, R_p , to describe the sedimentation of TP.

$$[\text{TP}] = J_T (1 - R_p) / Q$$

$$\text{or } [\text{TP}] = L (1 - R_p) / q_s$$

Many alterations have been proposed to this simple mass balance model and several improvements were incorporated in this study, following the approach that the best model is the simplest one that makes accurate predictions. Mass balance models which are semi-empirical, such as these, may best predict TP concentration on an annual basis or at least TP concentrations representing most of the year. Because one generally wishes to predict trophic status parameters during the ice-free period, the mass balance model was applied to predict mean ice-free concentrations of TP. To allow the ice-free period model to predict TP concentrations in other time periods, the ice-free TP concentrations were related (Table 48) to values of TP in other time periods (spring overturn, summer stratified, fall overturn, winter).

Differences in TP concentrations between vertical depth zones were observed in some lakes (Table 49). However, Snodgrass and Dillon (1983) compared one and two-layer mass balance models using a data set collected in Muskoka-Haliburton (Dillon 1974) and found that the one-layer model provided equally accurate predictions of TP concentration as did the two-layer model. Based on these results, the simple model described in equation 19 was not modified to account for thermal stratification.

Inherent in this simple model is the assumption that the concentration of TP in the lake and in the lake outlet are equal. Using data from the "A" lakes, the following modification was developed (equation 26).

$$[\text{TP}]_{\text{OUT}} = 0.956 [\text{TP}]_{\text{IF}}$$

Employing the above model revisions, the simplest version of the TP mass balance model (equation 27) was formulated as:

$$[\text{TP}] = L (1 - R_p) / 0.956 q_s$$

This version of the model was termed LCS-1. Alternatively, employing information on sedimentation loss of TP derived from whole-lake Pb burdens, Pb-210 accumula-

Table 71. Stepwise regression models for the prediction of gross annual export of total phosphorus ($\text{mg m}^{-2} \text{yr}^{-1}$). Independent variables and their units are listed in Table 70. All types of independent variables were used in model A whereas only geology and land use variables were used in model B.

Model	Step 1	Step 2	Step 3	Step 4	Step 5	Constant	SE	p
A ¹	0.38V27	0.012V2	-0.33V8	-0.29V28	-23.9V43	1.43	4.33	0.001
B ²	0.46V27	-0.27V8	-0.28V28	0.053V29	-	5.99	4.53	0.001

¹ $r^2 = 0.510$; $r_1^2 = 0.375$; $r_2^2 = 0.052$; $r_3^2 = 0.040$; $r_4^2 = 0.027$; $r_5^2 = 0.014$

² $r^2 = 0.456$; $r_1^2 = 0.375$; $r_2^2 = 0.030$; $r_3^2 = 0.032$; $r_4^2 = 0.019$

tion rates and TP concentrations in the sediments, the model was formulated as:

$$[TP] = (L - S) / 0.956 q_s$$

This version of the model was termed LCS-2.

Empirical calibration of TP retention (R_p) with data collected in oligotrophic Precambrian Shield lakes (Dillon and Kirchner 1975a) indicated that the settling velocity (v) averaged 12.4 m yr^{-1} . Employing this value, the LCS-1 model was tested on undeveloped Jerry Lake and Red Chalk Lake (main basin). Predicted TP concentrations agreed to within 0.1 mg m^{-3} with observed mean ice-free values. The TP concentrations predicted by LCS-2 did not agree as closely with the observed data for either Jerry Lake (10.6 mg m^{-3} vs 9.3 mg m^{-3} respectively) or the main basin of Red Chalk Lake.

Calibration of TP retention for the undeveloped east basin of Red Chalk Lake indicated that a settling velocity of 7.2 m yr^{-1} would be appropriate for lakes with anoxic hypolimnia. Therefore, the LCS-1 model was revised as follows:

$$[TP]_{IF} = L(1 - R_p) / 0.956 q_s$$

where: $R_p = 12.4 / (12.4 + q_s)$ for oxic lakes

$R_p = 7.2 / (7.2 + q_s)$ for lakes with anoxic hypolimnia

D. PHOSPHORUS, CHLOROPHYLL *a*, PHOTOPLANKTON AND WATER CLARITY

1. TOTAL PHOSPHORUS - CHLOROPHYLL *a* RELATIONSHIP

Chlorophyll *a*, the principal photosynthetic pigment in most phytoplankton characteristic of oligo- and mesotrophic soft-water lakes, is widely accepted as a surrogate for algal biomass. Although the chlorophyll *a* content per unit of cell mass (or volume) may be highly variable (Nicholls and Dillon 1978), the ease of measurement combined with the fact that chlorophyll *a* represents a readily visible indication of trophic status, has resulted in the frequent use of chlorophyll *a* measurements rather than direct phytoplankton biomass measurements in limnological studies. For these reasons, chlorophyll *a* is used as a primary measure of trophic status.

A relationship for prediction of chlorophyll *a* concentration in lakes was first proposed by Dillon and Rigler (1974) who combined a global data set with information collected for 19 lakes in southern Ontario:

$$\log \text{CHL } a_{SS} = 1.45 \log \text{TP}_{SO} - 1.14 \quad (37)$$

This relationship was based on the observation made by Sakamoto (1966) that TP and chlorophyll concentrations were highly correlated in a set of Japanese lakes with $\text{TN:TP} > 12$ (by weight). Since Sakamoto noted that lakes with low TN:TP (< 10) demonstrated better relationships between chlorophyll and TN than TP, Dillon and Rigler (1974) specified that the predictive relationship (equation 37) be used only in those cases where $\text{TN:TP} > 12$.

In the subsequent decade, almost 60 predictive relationships relating chlorophyll *a* and nutrients have been reported (some are reviewed by Nicholls and Dillon 1978 and by Straškraba 1978). The many differences in these

relationships are attributable to three facts:

(a) proponents of the various relationships have not been consistent with respect to analytical methods (chlorophyll *a*, chlorophyll *a + b + c*, and chlorophyll *a* corrected for phalopigments have all been used, often combined in a single relationship), sampling methodologies (volume-weighted vs. not volume-weighted samples, epilimnetic vs. euphotic zone vs. whole-lake samples), sampling times (annual vs. ice-free vs. stratification period), etc.,

(b) the chlorophyll content of algal cells may, as previously indicated, be quite variable. Chlorophyll may therefore only approximate algal biomass,

(c) the criterion that, for use of the TP-chlorophyll *a* relationship, the TN:TP ratio must be greater than 12 has often been ignored.

When model calibration has been restricted to consistent data sets and the original qualification concerning the TN:TP ratio has been observed, the results have been similar, supporting the premise that the TP-chlorophyll *a* relationship has considerable generality and predictive capability for north-temperate lakes.

Many of these relationships are not linear but predict that chlorophyll concentration increases faster than TP concentration. This has been explained by the fact that, as the TP input to lakes increases, the likelihood of anthropogenic contamination (almost invariably with readily biologically available phosphate-P) also increases. Therefore the proportion of the TP that is readily available for algal growth increases.

As a result of the many new relationships, significant improvements in predictability of chlorophyll *a* have been made, particularly by Smith and Shapiro (1981) and Smith (1982a, 1982b). These authors found (using global data sets with a wide variety of methodologies incorporated) that the chlorophyll *a* content (ice-free season) was dependent on the TN:TP ratio as well as the TP concentration of the lake itself:

$$\log \text{CHL } a_{IF} = 1.55 (\log \text{TP} - \log (6.404 / [0.0204 \text{ TN/TP} + 0.334])) \quad (38)$$

As the TN:TP ratio increases, its significance in predicting chlorophyll *a* decreases, such that if $\text{TN:TP} > 35$, the role of nitrogen is insignificant. Subsequently, a revised relationship that produced a family of chlorophyll *a* - TP plots depending on the TN:TP ratio was proposed:

$$\log \text{CHL } a_{IF} = 0.653 \log \text{TP} + 0.548 \log \text{TN} - 1.517 \quad (39)$$

These models can be used for lakes with any TN:TP ratio, removing one of the qualifiers of the early relationships. Subsequently, Janus and Vollenweider (1981) and Canfield (1982) also demonstrated the importance of nitrogen; multivariate relationships including both TP and TN produced improved predictive capability in each case.

The TN:TP ratios of the study lakes are shown in Table 72. In all cases, they were > 20 (usually > 30), ranging up to ≈ 50 in both the ice-free and summer stratification periods. It was therefore expected that nitrogen would not play a major role in determining chlorophyll concentrations.

¹ Sakamoto's analytical method yields chlorophyll measurements that are greater than chlorophyll *a* but less than chlorophyll *a + b + c*.

Table 72. TN/TP (by weight¹) averaged over the four-year (1976-79) period during the summer stratification (SS) and ice-free (IF) periods.

Lake	(N/P) _{SS}	(N/P) _{IF}
Blue Chalk	29.6	30.3
Chub	28.5	28.2
Dickie	24.0	26.7
Harp	38.5	39.9
Jerry	40.4	41.1
Red Chalk Main	51.8	52.3
Red Chalk East	42.6	44.1
Basshaunt	38.9	45.6
Bigwind	40.1	41.9
Buck	39.0	42.7
Crosson	30.5	31.1
Glen	21.6	24.7
Gullfeather	32.2	34.7
Little Clear	33.4	36.3
Solitaire	38.0	44.8
Walker	44.6	49.8

¹ Multiply by 2.21 for conversion to gram-atomic ratio

All of the aforementioned models were developed using data that covered a very wide range in parameter values; for example, Smith's most recent model (1982b) was developed from a data set (127 north-temperate lakes) with a range in chlorophyll *a* and TP concentrations of = three orders of magnitude. The study lakes, and indeed all of the lakes in the Precambrian portion of southern Ontario with the exception of a few eutrophied lakes, span a much narrower range in trophic status. The range in chlorophyll *a* concentrations in either the ice-free or summer stratification periods in the "A" and "B" lakes considering individual years is only 1.4 – 6.0 µg L⁻¹ (Table 73). The four-year mean values (Table 74) vary by only a factor of 3 (1.8 – 5.5 µg L⁻¹), a range 300 times smaller than that in Smith's (1982a) or Dillon and Rigler's (1974) model. The variation in chlorophyll *a* content between years was not large, with the average coefficient of varia-

tion being 20% (range 6-36%, 6-44%) for both the ice-free and summer stratification periods. Mean concentrations for the four-year period were, therefore, used in subsequent model development.

Models for prediction of mean chlorophyll *a* concentration during both the ice-free and summer stratification periods were developed using the stepwise regression procedure. Independent variables that were available for selection in addition to TP included TN, several individual nitrogen forms, and the TN:TP ratio (because of the findings of Smith and others), pH (indicative of degree of acidification of the lakes), colour and DOC (measures of both the natural acidity of the lakes and the dissolved organic content of the lakes) and lake morphometry (Table 75). For any lake the concentration of each

Table 74. Mean chlorophyll *a* concentration (µg L⁻¹) for the summer stratification (Chl *a*_{SS}) and ice-free periods (Chl *a*_{IF}). Coefficient of variation (c.v.) between years is expressed as %. Values are calculated as the mean of 4 annual means for the years 1976 to 1980.

Lake	Chl <i>a</i> _{SS}	(c.v.)	Chl <i>a</i> _{IF}	(c.v.)
Blue Chalk	1.83	(6)	1.82	(6)
Chub	3.01	(36)	3.01	(40)
Dickie	5.45	(9)	5.31	(12)
Harp	2.92	(9)	2.75	(10)
Jerry	3.16	(29)	3.09	(29)
Red Chalk Main	2.05	(9)	1.99	(8)
Red Chalk East	2.52	(13)	2.49	(12)
Basshaunt	1.78	(32)	1.74	(29)
Bigwind	2.64	(18)	2.47	(19)
Buck	2.03	(34)	2.27	(44)
Crosson	2.50	(31)	2.27	(28)
Glen	2.61	(30)	3.13	(10)
Gullfeather	4.82	(17)	4.56	(14)
Little Clear	2.32	(11)	2.15	(17)
Solitaire	1.80	(17)	1.72	(20)
Walker	2.39	(23)	2.27	(18)
	mean (20)		mean (20)	

Table 73. Mean annual chlorophyll *a* concentration (µg L⁻¹) for the summer stratification (Chl *a*_{SS}) and ice-free periods (Chl *a*_{IF}) for the years 1976 to 1979.

Lake	Chl <i>a</i> _{SS}				Chl <i>a</i> _{IF}			
	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	1.88	1.89	1.68	1.87	1.87	1.85	1.67	1.89
Chub	3.33	1.98	4.41	2.34	3.14	1.99	4.65	2.26
Dickie	5.97	5.03	5.10	5.73	4.51	5.85	5.79	5.10
Harp	3.01	2.92	2.55	3.21	2.71	2.78	2.42	3.08
Jerry	2.03	2.85	3.70	4.06	2.05	2.73	3.48	4.10
Red Chalk Main	2.26	1.81	2.12	2.01	2.10	1.75	2.09	2.01
Red Chalk East	— ¹	2.26	2.89	2.42	— ¹	2.24	2.83	2.39
Basshaunt	2.00	1.00	1.80	2.32	1.84	1.07	1.78	2.29
Bigwind	2.07	2.50	3.18	2.82	1.84	2.64	2.94	2.46
Buck	2.05	1.32	1.79	2.98	1.94	1.43	1.97	3.74
Crosson	3.50	1.80	2.72	1.98	3.04	1.66	2.55	1.85
Glen	1.78	— ¹	2.72	3.33	3.05	3.00	2.90	3.57
Gullfeather	4.94	4.48	5.92	3.96	4.67	4.99	4.93	3.66
Little Clear	2.41	2.50	2.43	1.94	2.56	2.06	2.31	1.69
Solitaire	2.18	1.82	1.42	1.78	2.19	1.77	1.40	1.51
Walker	2.33	1.78	2.35	3.12	2.28	1.80	2.20	2.81

¹ no data

of the chemical variables can be predicted (TP) using the methodologies outlined in this report or easily measured (other than TP) during the appropriate time period. Any model must, of course, be applied for predictive purposes with extreme caution to cases where the independent parameters are less than or greater than the range used in model construction. These ranges are also shown in Table 75.

The most significant correlations between mean chlorophyll *a* concentration in the study lakes ("A" and "B" lakes) and independent chemical variables are shown in Table 76. For both the ice-free and summer stratification periods, TP concentration (epilimnetic) was the strongest correlate; however, strong correlations also existed with

Table 75. Independent variables used in the generation of chlorophyll *a* models. For each parameter, epilimnetic (EP), metalimnetic (ME), summer stratification (SS), ice-free period (IF), ice-covered period (IC), spring overturn (SO), fall overturn (FO), and annual average (AN) data were used. For TP volume-weighted epilimnetic plus metalimnetic (EM) data were also used. Ranges in the epilimnion, and the whole lake during the ice-free and annual periods are shown. Units are mg m⁻³ unless specified.

Parameter	Symbol	Range		
		EP	IF	AN
total phosphorus	TP	4.9-13.0	5.6-19.3	5.7-20.7
nitrate (+ nitrite) ¹	NO ₃	8.7-111	16-133	18-138
ammonium	NH ₄	9.0-20	12-147	14-158
total Kjeldahl nitrogen	TKN	190-310	210-440	210-460
total organic nitrogen (TKN - NH ₄)	TON	180-290	190-300	190-310
total inorganic nitrogen (NO ₃ + NH ₄)	TIN	19-127	35-190	40-220
total nitrogen (TKN + NO ₃)	TN	200-410	220-480	230-520
total nitrogen/total phosphorus	N/P	25.8-54.7	24.7-52.3	25.3-53.7
inorganic nitrogen/total phosphorus	I/P	2.6-9.8	4.7-17.9	5.7-18.4
pH (-)	PH	5.86-7.83	5.58-7.39	5.58-7.23
colour (Hazen units)	COL	<5.0-12.4	5.0-22.9	5.2-24.4
dissolved organic carbon (mg L ⁻¹)	DOC	2.3-5.8	2.7-6.7	2.8-6.9
lake area (10 ⁴ m ²)	A		10.9-124	
lake volume (10 ⁶ m ³)	V		0.75-16.4	
mean depth (m)	Z		4.8-16.7	
maximum depth (m)	ZM		12.0-40.0	

¹ nitrite was always undetectable when measured separately

other TP parameters (volume-weighted epilimnetic plus metalimnetic concentration), with several nitrogen parameters and with DOC and colour. Although the range in chlorophyll *a* concentration among lakes was extremely small (3-fold), TP (epilimnetic) concentration explained 77% of the variance in each time period.

Stepwise regression models for the two time periods are shown in Table 77. The second step in each model (highly significant model improvement each time) was the entry of the TN:TP ratio (epilimnetic). Although this is consistent with the findings of Smith and Shapiro (1981) and Smith (1982a), it is surprising because the ranges in both chlorophyll *a* and TN:TP were small (three-fold and two-fold, respectively). Other nitrogen parameters were selected in subsequent steps, again emphasizing that nitrogen and phosphorus, not phosphorus alone, determine chlorophyll *a* standing stock. The complete models explained 92 and 88% of the variance in chlorophyll *a* in the ice-free and summer stratification periods respectively.

For predictive purposes, truncated (2-step) models are presented in Table 78. Using only TP (epilimnetic) and the TN:TP ratio (epilimnetic), 82 and 84% of the variance in chlorophyll *a* in the two time periods is explained. The resultant models are:

$$\text{Chl } a_{SS} = 0.51 \text{ TP}_{EP} + 0.059 (\text{N/P})_{EP} - 3.35 \quad (40)$$

$$\text{Chl } a_{IF} = 0.48 \text{ TP}_{EP} + 0.060 (\text{N/P})_{EP} - 3.14 \quad (41)$$

To facilitate use of the models by eliminating the need for measurement of TN:TP in the epilimnion over the entire summer stratification period for a particular lake, the relationship

Table 76. Correlations between mean chlorophyll *a* concentration (euphotic zone) during summer stratification (SS) and the ice-free period (IF) and independent chemical variables (listed in Table 75).

Chl <i>a</i> _{SS}			Chl <i>a</i> _{IF}		
variable	r ¹	F ²	variable	r ¹	F ²
TP _{EP}	0.878	47.0	TP _{EP}	0.870	43.4
DOC _{AN}	0.854	37.7	TP _{EM}	0.868	42.8
TP _{EM}	0.852	37.0	TON _{EP}	0.832	31.4
TN _{EP}	0.806	25.9	TKN _{EP}	0.826	30.0
TON _{EP}	0.797	24.3	DOC _{AN}	0.817	28.2
TKN _{EP}	0.791	23.4	TN _{EP}	0.806	25.9
COL _{IC}	0.770	20.4			

¹ correlation coefficient

² partial F-value

Table 77. Stepwise regression models for the prediction of Chl *a* concentration in the euphotic zone during the summer stratification (SS) and ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Step 3	Step 4	Constant	p	SE
(Chl <i>a</i>) _{SS} ¹	0.59 TP _{EP}	0.082 (N/P) _{EP}	-0.0063 TIN _{IF}	-	-4.18	<0.001	0.394
(Chl <i>a</i>) _{IF} ²	0.83 TP _{EP}	0.12 (N/P) _{EP}	-0.0018 TIN _{EP}	-0.0076 TON _{IC}	-5.56	<0.001	0.330

¹ r² = 0.882; r₁² = 0.771; r₂² = 0.064; r₃² = 0.047

² r² = 0.919; r₁² = 0.756; r₂² = 0.064; r₃² = 0.061; r₄² = 0.038

$$(TN/TP)_{EP} = 0.43 (TN/TP)_{FO} + 19.4 \quad (42)$$

may be used. This relationship ($r^2 = 0.58$; $p < 0.05$) requires measurement of the TN:TP ratio only at fall overturn, i.e., one or two measurements. A similar relationship based on the spring overturn period was less useful.

Table 78. Stepwise regression models truncated after the second step for the prediction of mean Chl *a* concentration in the euphotic zone during summer stratification (SS) and ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Constant	p	SE
(Chl <i>a</i>) _{SS} ¹	0.51 TP _{EP}	0.059 (N/P) _{EP}	-3.35	<0.001	0.451
(Chl <i>a</i>) _{IF} ¹	0.48 TP _{EP}	0.060 (N/P) _{EP}	-3.14	<0.001	0.453

¹ $r^2 = 0.835$; $r_1^2 = 0.771$; $r_2^2 = 0.064$

² $r^2 = 0.820$; $r_1^2 = 0.756$; $r_2^2 = 0.064$

2. PHYTOPLANKTON COMPOSITION, BIOMASS AND TOTAL P

It is generally assumed that increases in nutrient levels in lake waters will increase total phytoplankton biomass (PB) and change the taxonomic composition of the phytoplankton community.

A total of 177 phytoplankton genera were observed in the 16 lakes in the four years (Nakamoto et al. 1983). The highest proportional contribution was from the green algae, Chlorophyceae (32 genera). The total number of genera in each lake (four-year averages) ranged from 38 for Glen Lake to 82 for Red Chalk Lake East and was not correlated with TP (after correcting for numbers of samples examined).

Based on the genus data (177 genera $\times$ 16 lakes $\times$ 4 years) a clustering analysis of dissimilarity coefficients among the lakes showed much greater year-to-year than lake-to-lake differences. Lakes in 1976 were alike, but different from the lakes in 1977-78, and different again from the lakes in 1979. Some of this variability might have been related to the rate of warming of the lakewater in early summer. The local air temperatures during April-May (Muskoka airport) were most similar during 1977-78 (98% similarity) compared with 1976-79 (95% similarity).

Based on both the total phytoplankton biomass (PB) and the proportional contribution to PB of the seven dominant algal classes, there were marked differences in calculated percentage similarities among the lakes. Dickie and Gullfeather lakes were similar to each other but unlike any of the other lakes. Jerry Lake also was unlike most of the other lakes and was similar to only Red Chalk Main, Glen and Harp Lakes at a level greater than 80%. In contrast, Crossson Lake had the highest average level of similarity with all other lakes and showed similarity levels exceeding 80% with all lakes except Jerry, Gullfeather and Dickie (Figure 19).

Noxious blooms of blue-green algae have been reported in hard-water (Nicholls et al. 1983) and in nutrient-enriched soft-water lakes in Ontario (Dillon et al. 1978,

Schindler and Fee 1974). Such blooms may render water toxic to animals, farm animals for example, that may normally consume the water, and also drastically impair aesthetic and recreational appeal of lakes.

The proportion of PB of the lakes contributed by blue-greens was very low (4-year ice-free period averages ranged from 0.1-4.1%), and was not significantly correlated with either TP or the ratio of TN:TP. Data from other Ontario lakes suggest that the significant occurrence ($>0.2 \text{ mm}^3/\text{L}$) of blue-green bloom-forming algae is not likely in lakewaters with TN:TP mass ratios greater than 25. The TN:TP values for the study lakes (4-year ice-free period means) ranged between 25 and 52 (Table 72).

Composition of phytoplankton communities may be influenced by light. Hence, differences in community structure among the LCS lakes might be related to differences in relative illuminance of their epilimnia. The effective light climate (*I* in Table 79) was apparently an

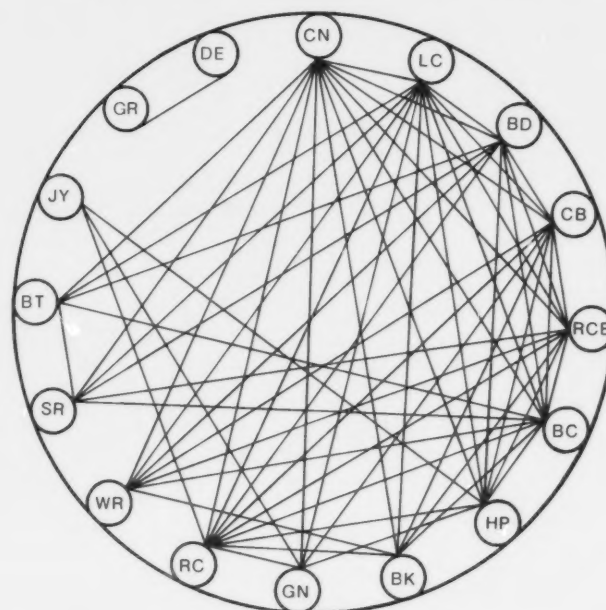


Figure 19 Similarity of phytoplankton communities among the 16 Muskoka-Haliburton lakes based on a 1:1 weighting of average phytoplankton biomass (1976-79) and the proportionality of algal class composition. Lakes joined by a straight line indicate a similarity $>80\%$. The lakes are arranged in clockwise order of highest (Crossson - CN) to lowest (Dickie - DE) average percentage similarity with all other lakes in the set.

- BC — Blue Chalk
- BD — Bigwind
- BK — Buck
- BT — Basshaunt
- CB — Chub
- CN — Crossson
- DE — Dickie
- GN — Glen
- GR — Gullfeather
- HP — Harp
- JY — Jerry
- LC — Little Clear
- RC — Red Chalk
- RCE — Red Chalk East
- SR — Solitaire
- WR — Walker

important factor in determining the composition of the lakes' phytoplankton. The importance (mg algae per mg TP) of diatoms and green algae in the clear water (high I) lakes and of the Chrysophyceae and Dinophyceae in the

Table 79. Average biomass of phytoplankton per unit total phosphorus (BP/TP) and average effective light climate ($\bar{I}$)¹ for the epilimnia of the "A" lakes during 1979.

Lake	PB/TP (mm ³ /g)		$\bar{I}$ (cal/cm ² .day)	
	June- July	Aug.- Sept.	June- July	Aug.- Sept.
Blue Chalk	0.12	0.072	130	65
Red Chalk - Main	0.098	0.126	140	70
Red Chalk - East	0.22	0.076	129	72
Chub	0.052	0.052	104	52
Dickie	0.128	0.151	87	44
Harp	0.118	0.119	112	61
Jerry	0.07	0.106	123	61
Mean $\pm$ St. D.	0.10 $\pm$ 0.03 ²	0.11 ⁵ $\pm$ 0.005 ²	118 $\pm$ 18 ³	61 $\pm$ 10 ³

$$\bar{I} = \frac{I_0^1 (1 - e^{-kz})}{kz}$$

where - I_0^1 is downwelling photosynthetically available radiation at the lake surface estimated as 41% of total downwelling radiation (200-3000nm) measuring in the field.

- k is the apparent vertical extinction coefficient of downwelling photosynthetically available radiation. This was estimated as 1.45/Secchi transparency depth (Walker 1978)

- z is measured epilimnetic depth

² not significantly different (t-test; $p > 0.05$)

³ significantly different (t-test; $p < 0.05$)

coloured lakes (low $\bar{I}$) confirms the experimental evidence of Wall and Briand (1979) which showed that diatoms and green algae preferred higher light intensities than certain flagellates.

The equation relating Cryptophyceae plus Dinophyceae importance (units as above) to $\bar{I}$ (cal/cm².day) was:

$$\text{Crypto} + \text{Dino} = -0.35 \bar{I} + 43.7 \quad (r = 0.76; P < 0.01) \quad (43)$$

and the equation relating *Cyclotella* (a diatom) and Chlorophyceae to $\bar{I}$ (cal/cm².day) was:

$$\text{Cyclotella} + \text{Chloro} = 0.42^{0.041} \bar{I} \quad (r = 0.62; P < 0.05) \quad (44)$$

The mean phytoplankton biomass measured in the ice-free and summer stratification periods in the "A" and "B" lakes is shown in Table 80. Like chlorophyll *a*, the range in phytoplankton biomass among lakes was very small. The four-year mean concentrations (Table 80) ranged from only 0.32 mg L⁻¹ (Basshaunt Lake) to 1.08 mg L⁻¹ (Dickie Lake) during summer stratification. However, with the average coefficient of variation between years being 30% (Table 81), the differences were greater than for chlorophyll *a*.

Analysis of variance and least significant difference tests showed that the average ice-free season estimate of phytoplankton biomass of all lakes for any one of the four years was not significantly different from that of any other year. The values were 0.57, 0.54, 0.65 and 0.58 mg³/L for 1976, 1977, 1978 and 1979, respectively.

Unlike chlorophyll *a*, phytoplankton biomass has been infrequently used as a measure of the trophic status of lakes. This may be largely attributed to the difficulty in measurement of biomass compared to chlorophyll *a*.

Few predictive models for phytoplankton biomass have been proposed. Nicholls and Dillon (1978) demonstrated that variance in a TP-chlorophyll *a* model for relatively hard-water lakes in southern Ontario could be substantially reduced if chlorophyll *a* was replaced by phytoplankton biomass. Kalff and Knoechel (1978) developed a predictive relationship with TP concentration

Table 80. Mean phytoplankton biomass (mg L⁻¹) for the summer stratification (PB_{SS}) and ice-free (PB_{IF}) periods for the years 1976 to 1979.

Lake	PB _{SS}				PB _{IF}			
	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	0.34	0.37	0.42	0.37	0.33	0.36	0.39	0.40
Chub	0.79	0.36	0.59	0.43	0.83	0.38	0.77	0.42
Dickie	0.86	0.72	0.92	1.81	0.66	0.90	1.11	1.62
Harp	0.75	0.66	0.69	0.66	0.72	0.64	0.62	0.64
Jerry	0.35	0.76	1.41	0.66	0.38	0.75	1.36	0.66
Red Chalk Main	0.96	0.56	0.51	0.42	0.92	0.54	0.48	0.41
Red Chalk East	- ¹	0.53	0.49	0.52	- ¹	0.50	0.47	0.50
Basshaunt	0.21	- ¹	0.40	0.36	0.21	- ¹	0.37	0.43
Bigwind	0.34	0.78	0.54	0.32	0.31	0.72	0.52	0.28
Buck	0.73	0.18	0.43	0.43	0.73	0.24	0.44	0.59
Crosson	0.91	0.25	0.78	0.27	0.67	0.22	0.69	0.25
Glen	0.50	- ¹	1.07	0.87	0.50	- ¹	1.07	0.72
Gullfeather	0.87	0.85	1.04	0.92	0.93	1.18	1.01	0.83
Little Clear	0.53	0.49	0.23	0.62	0.54	0.51	0.23	0.52
Solitaire	0.30	0.37	0.33	0.37	0.30	0.38	0.30	0.34
Walker	0.42	0.27	0.66	0.66	0.35	0.31	0.57	0.58

¹ no data

using a small global data set. Schindler et al. (1978) found very strong correlations between algal biomass (or volume) and TP (epilimnetic or whole-lake) concentration, as well as with several functions combining TP load, lake morphometry and lake water balance, for a number of lakes in the Experimental Lakes Area, Ontario.

Correlations between phytoplankton biomass and chemical variables are shown in Table 82. Total organic nitrogen (epilimnetic) and TP (epilimnetic plus metalimnetic) were the best correlates of phytoplankton biomass during the summer stratification and ice-free periods respectively; however, in each case, other phosphorus and nitrogen parameters were almost equally highly correlated. Using stepwise regression analyses with F ratios to add or delete independent variables set at 90%,

Table 81. Mean phytoplankton biomass (mg L^{-1}) for the summer stratification (PB_{SS}) and ice-free (PB_{IF}) periods. Coefficient of variation (c.v.) between years is expressed as %. Values are calculated as the mean of 4 annual means for the years 1976 to 1980.

Lake	PB _{SS}	(c.v.)	PB _{IF}	(c.v.)
Blue Chalk	0.38	(9)	0.37	(9)
Chub	0.54	(35)	0.60	(39)
Dickie	1.08	(46)	1.07	(38)
Harp	0.69	(6)	0.66	(7)
Jerry	0.80	(56)	0.79	(52)
Red Chalk Main	0.61	(39)	0.59	(39)
Red Chalk East	0.51	(4)	0.49	(4)
Basshaunt	0.32	(31)	0.34	(33)
Bigwind	0.50	(43)	0.46	(45)
Buck	0.44	(51)	0.50	(42)
Crosson	0.55	(62)	0.46	(56)
Glen	0.81	(36)	0.76	(38)
Gullfeather	0.92	(9)	0.99	(15)
Little Clear	0.47	(36)	0.45	(33)
Solitaire	0.34	(10)	0.33	(12)
Walker	0.50	(38)	0.45	(32)
	mean (32)		mean (31)	

one-variable equations were formulated (Table 83); there were no significant additions at the second step. These models explained 62 and 63% of the variance in phytoplankton biomass in the summer stratification and ice-free periods, substantially less than was explained for chlorophyll *a*. This result was unexpected since chlorophyll *a* is considered to be only an approximation of algal biomass, the parameter that is anticipated to be highly correlated with nutrient levels.

Of the components of the algal biomass, the diatoms (Bacillariophyceae) comprise a significant portion in many soft-water oligotrophic and mesotrophic lakes.

Table 82. Correlations between mean total phytoplankton biomass in the euphotic zone during summer stratification (SS) and the ice-free (IF) period and independent chemical variables (listed in Table 75).

variable	PB _{SS}		variable	PB _{IF}	
	r ¹	F ²		r ¹	F ²
TON _{EP}	0.789	22.6	TP _{EM}	0.800	24.1
TP _{EM}	0.777	21.3	TON _{EP}	0.784	22.3
TKN _{EP}	0.773	20.7	TKN _{EP}	0.772	20.6
TN _{EP}	0.748	17.8	TN _{EP}	0.771	20.5
TP _{EP}	0.729	15.9	TP _{EP}	0.767	20.0

¹ correlation coefficient

² partial F-value

Table 83. Stepwise regression models for the prediction of mean total phytoplankton biomass (PB) in the euphotic zone during the summer stratification (SS) and ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Constant	p	SE
PB _{SS} ¹	0.0052 TON _{EP}	-0.64	<0.001	0.137
PB _{IF} ²	0.068 TP _{EM}	-0.0044	<0.001	0.139

¹ r² = 0.617

² r² = 0.633

Table 84. Mean corrected phytoplankton biomass (less diatom biomass) (mg L^{-1}) for the summer stratification (CPB_{SS}) and ice-free (CPB_{IF}) periods for the years 1976 to 1979.

Lake	CPB _{SS}				CPB _{IF}			
	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	0.17	0.25	0.25	0.13	0.16	0.24	0.22	0.18
Chub	0.35	0.29	0.44	0.39	0.41	0.30	0.70	0.38
Dickie	0.83	0.69	0.73	1.42	0.56	0.82	0.87	1.28
Harp	0.33	0.29	0.43	0.36	0.33	0.29	0.38	0.35
Jerry	0.19	0.28	0.35	0.55	0.23	0.28	0.32	0.54
Red Chalk Main	0.20	0.15	0.29	0.20	0.22	0.16	0.29	0.40
Red Chalk East	- ¹	0.26	0.44	0.40	- ¹	0.25	0.42	0.39
Basshaunt	0.20	- ¹	0.18	0.21	0.20	- ¹	0.18	0.33
Bigwind	0.18	0.18	0.40	0.21	0.16	0.25	0.38	0.20
Buck	0.18	0.17	0.23	0.20	0.20	0.21	0.28	0.41
Crosson	0.36	0.20	0.33	0.20	0.30	0.19	0.29	0.20
Glen	0.34	- ¹	0.63	0.43	0.34	- ¹	0.52	0.40
Gullfeather	0.69	0.75	0.89	0.85	0.68	0.87	0.85	0.75
Little Clear	0.22	0.26	0.18	0.28	0.27	0.32	0.20	0.27
Solitaire	0.15	0.25	0.23	0.22	0.15	0.27	0.22	0.22
Walker	0.26	0.23	0.38	0.62	0.23	0.27	0.40	0.55

¹ no data

Since a large proportion of many diatom species consists of a vacuole (empty space) while the cytoplasmic portion of the cell is highly variable, and since it is often difficult to distinguish between living and dead diatoms, the total phytoplankton biomass excluding diatoms (called the "corrected phytoplankton biomass") was calculated (Table 84). In many cases, these values were substantially lower than the total phytoplankton biomass (e.g., Red Chalk Lake, Main Basin - 0.21 vs. 0.61 mg L⁻¹ in the summer stratification period). The variability between years (Table 85) was comparable (coefficient of variation 27 and 28% in the two time periods) to that of total phytoplankton biomass.

The correlations between the corrected phytoplankton biomass and nutrient parameters (Table 86) were much better than when total phytoplankton biomass was used (Table 82). In both time periods the epilimnetic and combined epilimnetic and metalimnetic TP concentrations were the best correlates, with several nitrogen fractions and DOC also strongly correlated. In neither instance was pH significantly correlated.

Stepwise regression models were constructed for corrected phytoplankton biomass using the epilimnetic TP concentration as the first step in each time period for consistency (Table 87). As was the case for chlorophyll *a*, the TN:TP ratio was the second parameter added to each

Table 85. Mean corrected phytoplankton biomass (less diatom biomass), for the summer stratification (CPB_{SS}) and ice-free (CPB_{IF}) periods. Coefficient of variation (c.v.) between years is expressed as %. Values are calculated as the mean of 4 annual means for the years 1976 to 1980.

Lake	CPB _{SS}	(c.v.)	CPB _{IF}	(c.v.)
Blue Chalk	0.20	(30)	0.20	(18)
Chub	0.37	(17)	0.45	(39)
Dickie	0.92	(37)	0.88	(34)
Harp	0.35	(17)	0.34	(11)
Jerry	0.34	(45)	0.34	(40)
Red Chalk Main	0.21	(28)	0.27	(38)
Red Chalk East	0.37	(26)	0.35	(26)
Basshaunt	0.20	(7)	0.23	(35)
Bigwind	0.24	(44)	0.25	(38)
Buck	0.20	(13)	0.28	(35)
Crosson	0.27	(31)	0.25	(23)
Glen	0.47	(32)	0.42	(22)
Guilfeather	0.80	(11)	0.79	(11)
Little Clear	0.24	(18)	0.27	(18)
Solitaire	0.21	(21)	0.22	(22)
Walker	0.37	(48)	0.36	(40)
	mean (27)		mean (28)	

Table 87. Stepwise regression models for the prediction of mean phytoplankton biomass excluding diatoms (CPB) in the euphotic zone during summer stratification (SS) and the ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Step 3	Step 4	Step 5	Constant	p	SE
CPB _{SS} ^{1,3}	0.31 TP _{EP}	0.070 (N/P) _{EP}	-0.10 (I/P) _{EP}	-0.0066 TKN _{EP}	0.76 pH _{IF}	-2.97	<0.001	0.039
CPB _{IF} ²	0.25 TP _{EP}	0.039 (N/P) _{EP}	-0.0053 TN _{EP}	0.080 pH _{EP}	-	-2.10	<0.001	0.059

¹ r² = 0.978; r₁² = 0.718; r₂² = 0.101; r₃² = 0.077; r₄² = 0.037; r₅² = 0.027

² r² = 0.933; r₁² = 0.766; r₂² = 0.075; r₃² = 0.070; r₄² = 0.022

³ two additional significant steps (NO_{2AN}, TON_{IC}) contributed slightly to r²

model. The addition of this parameter resulted in very significantly increased predictive capability of the models. With all variables explaining significant amounts of variance of CPB (five in the summer stratification period, four in the ice-free period), the regression models explained 98 and 93% of the variance in corrected phytoplankton biomass, despite the narrow range in biomass. Truncated models (Table 88) which require less complete data accounted for 82 and 84% of the variance in corrected phytoplankton biomass (CPB).

The resultant models are:

$$CPB_{SS} = 0.108TP_{EP} + 0.015(TN/TP)_{EP} - 1.05 \quad (45)$$

$$CPB_{IF} = 0.098TP_{EP} + 0.012(TN/TP)_{EP} - 0.84 \quad (46)$$

In summary, phytoplankton biomass excluding diatoms can be predicted much more successfully than total phytoplankton biomass, and slightly more successfully than chlorophyll *a*. The phytoplankton concentration (either biomass or chlorophyll *a*) in these lakes is controlled by nutrient, particularly TP, concentrations, not by the pH of the lakewater.

Table 86. Correlations between mean phytoplankton biomass excluding diatoms (CPB) in the euphotic zone during summer stratification (SS) and the ice-free (IF) periods. Independent variables and their units are listed in Table 75.

CPB _{SS}			CPB _{IF}		
variable	r ¹	F ²	variable	r ¹	F ²
TP _{EM}	0.862	40.6	TP _{EP}	0.875	45.9
TP _{EP}	0.847	35.6	TP _{EM}	0.869	43.3
TON _{EP}	0.806	26.0	DOC _{AN}	0.805	25.7
TKN _{EP}	0.804	25.6	TN _{EP}	0.799	24.7
TN _{EP}	0.803	25.4	TKN _{EP}	0.785	22.4
DOC _{AN}	0.780	21.8	TON _{EP}	0.785	22.4
			COL _{IC}	0.773	20.7

¹ correlation coefficient

² partial F-value

Table 88. Stepwise regression models truncated after the second step for the prediction of mean phytoplankton biomass excluding diatoms (CPB) in the euphotic zone during summer stratification (SS) and the ice-free (IF) periods. Independent variables and their units are listed in Table 75.

	Step 1	Step 2	Constant	p	SE
CPB _{SS} ¹	0.108 TP _{EP}	0.015 (N/P) _{EP}	-1.05	<0.001	0.097
CPB _{IF} ²	0.098 TP _{EP}	0.012 (N/P) _{EP}	-0.84	<0.001	0.084

¹ r² = 0.819; r₁² = 0.718; r₂² = 0.101

² r² = 0.841; r₁² = 0.766; r₂² = 0.075

3. EFFECTS OF ZOOPLANKTON BIOMASS ON PHOSPHORUS-PHYTOPLANKTON RELATIONSHIPS

Standing stocks of phytoplankton are determined by rates of algal growth, a phosphorus-dependent process, and by rates of loss or removal of algae from the water column. Zooplankton grazing may be the most important removal mechanism. Explicit consideration of differences among lakes in expressions of zooplankton community grazing pressure may, therefore, result in improvements in predictability of phytoplankton biomass.

Direct measurements of herbivorous zooplankton grazing or filtering rates were not made in the study lakes. As filtering rates (to be discussed in detail) of individual zooplankters are strongly correlated with animal size (Burns 1969), community filtering rates should be positively correlated with community biomass, the product of zooplankton community density (animals L⁻¹) and average zooplankton size. In this section whether predictions of water clarity or phytoplankton biomass in the study lakes may be improved by inclusion of total crustacean (B_Z) or herbivorous crustacean (B_H) zooplankton biomass in regression models is examined. Classification of animals as herbivores or carnivores followed Sprules (1980).

Zooplankton were enumerated in volume-weighted whole lake composite samples. Average biomass values generated from these samples are given in Table 89. Averages of B_Z for the ice-free season ranged from 27.3 to 88.4 mg m⁻³ (dry weight) among the "A" lakes from 1977 to 1979. The average for all lakes was 48.7 mg m⁻³. Among the "B" lakes, B_Z ranged from 20.3 to 125.1 mg m⁻³ with a mean of 64.1 mg m⁻³. B_H averaged 74% of B_Z.

B_Z and B_H were added to the set of independent variables used for model development. The stepwise regression procedure was then repeated using ice-free season and summer stratification season averages of Secchi depth, chlorophyll *a*, phytoplankton biomass, and corrected

phytoplankton biomass as dependent variables. Neither B_Z nor B_H explained a sufficient amount of the residual variance in any of the dependent variables to be included in any of the regression models.

The regression analyses were then repeated for all dependent variables excluding Secchi transparency, with first B_Z then B_H inserted as the second independent variable. Decision criteria for choice of the first independent variable were unchanged. In a second exercise, B_Z and B_H were also inserted as the third independent variable. In all cases, insertion of B_Z or B_H resulted in no significant (*p* > 0.10) reduction in residual variance in chlorophyll *a* or phytoplankton biomass (Table 90).

These analyses should not be interpreted as signifying that zooplankton are never important in any particular lake or at any particular time in controlling phytoplankton biomass. Recent changes in *Daphnia* populations in Lake Washington in the U.S.A., for example, have had substantial effects on the phytoplankton of the lake (Edmundson and Litt 1982). Nevertheless, the analyses do indicate that explicit consideration of average

Table 90. Examples of changes in coefficient of determination (*r*²) for regressions produced by selection of either B_Z or B_H as the second independent variable in multiple linear regression models. Note that the increase in *r*² is small (and not statistically significant) in each case.

Dependent Variable ¹	First Independent Variable		<i>r</i> ² with	
	Name	<i>r</i> ²	B _Z	B _H
Chlor <i>a</i> _{IF}	TP _{EP}	0.756	0.765	0.759
Chlor <i>a</i> _{SS}	TP _{EP}	0.771	0.795	0.787
PB _{IF}	TP _{EM}	0.633	0.693	0.682
PB _{SS}	TON _{EP}	0.617	0.618	0.617
CPB _{IF}	TP _{EP}	0.766	0.773	0.769
CPB _{SS}	TP _{EP}	0.718	0.719	0.721

¹ PB is total phytoplankton biomass; CPB is total phytoplankton biomass less diatom biomass.

Table 89. Ice-free season averages of total (B_Z) and herbivorous (B_H) crustacean zooplankton biomass in the study lakes for 1977, 1978 and 1979. Arithmetic means of 1977-1979 are also included.

Lake	1977		1978		1979		mean	
	B _Z	B _H	B _Z	B _H	B _Z	B _H	B _Z	B _H
Blue Chalk	44.7	36.3	56.0	39.4	69.9	51.5	56.9	42.4
Chub	47.6	30.4	49.1	26.6	44.8	36.6	47.2	31.2
Dickie	88.4	80.7	28.3	21.4	43.9	27.4	53.5	43.1
Harp	33.9	27.1	39.5	27.1	47.1	36.1	40.2	30.1
Jerry	27.3	18.1	43.2	26.4	73.7	44.3	48.1	29.6
Red Chalk Main	30.6	21.4	37.1	23.7	43.7	25.1	37.1	23.4
Red Chalk East	37.3	27.7	75.6	56.6	61.8	45.7	58.2	43.3
Basshaunt	—	—	46.9	27.0	60.0	46.1	53.5	36.5
Bigwind	41.1	35.5	50.4	42.2	55.0	45.6	48.8	41.1
Buck	20.7	14.7	34.0	26.1	42.7	36.7	32.5	25.8
Crosson	70.3	36.4	74.2	62.3	76.2	59.4	73.5	52.7
Glen	—	—	108.7	86.8	99.7	88.2	104.2	87.5
Gullfeather	36.6	29.9	125.1	93.3	123.5	99.1	95.1	74.1
Little Clear	65.6	49.5	83.8	56.8	63.4	55.8	70.9	54.0
Solitaire	36.0	23.5	47.2	32.4	67.7	51.8	50.3	36.0
Walker	20.3	17.0	60.4	51.0	64.8	50.6	48.5	39.5

total or herbivorous zooplankton biomass does not significantly improve the predictability of long-term average phytoplankton biomass for relatively nutrient poor lakes in the Muskoka-Haliburton area.

It is assumed for this analysis that zooplankton biomass was a good correlate of zooplankton community filtering rate. This assumption implies that filtering rates per unit biomass should be constant among species. As Egloff and Palmer (1974) indicated, this may not be true, the analyses were repeated using estimated community filtering rates.

Zooplankton filtering rates (FR) can be conceptualized as that amount of lake water (for individual zooplankters) or that proportion of the lake (for the zooplankton community) which is effectively swept clear of prey by the grazing activities of the animals in unit time. Typically, individual FR is expressed in units of mL animal⁻¹ d⁻¹ and community FR in units of mL of cleared lake water per L of lake water per day or in % (of the lake) d⁻¹.

FR was estimated for individual zooplankters as follows. Average lengths of all common herbivores were calculated from their measured average weights using literature length-weight relationships. FR was calculated from length-FR relationships developed by Chow (unpub. studies) for species commonly found in Ontario lakes. FR ranged from 0.28 mL animal⁻¹ d⁻¹ for *Eubosmina tubicen* to 24.09 mL animal⁻¹ d⁻¹ for *Daphnia rosea*. The mean FR was 5.3 mL animal⁻¹ d⁻¹ (Table 91). This calculated value

is similar to measured values in Ontario lakes (Haney 1973).

It was previously assumed that B_z and B_h were excellent correlates of community FR, i.e., that FR/W, where W is the average weight (μg animal⁻¹) of an individual of a taxon, would be identical for all taxa. The calculated values of FR/W for each taxon ranged over an order of magnitude, however, from 0.7 mL μg⁻¹ d⁻¹ or less for *B. longirostris*, *Chydorus* spp., *E. longispina* and *E. tubicen* to about 5 mL μg⁻¹ d⁻¹ for nauplii (Table 91). This suggests that the assumption of constant FR/W might (recall FR was estimated not measured) have been in error and that B_z and B_h may not be good estimators of community FR, as species composition of zooplankton does vary among the study lakes.

Estimates of community FR were, therefore, generated for the ice-free season for each lake. FR was taken to be the sum for all taxa of the product of FR for individuals of each taxon (Table 92) and the densities of those animals (animals L⁻¹). Calculated community FR ranged from 1.96% d⁻¹ in 1977 in Buck Lake to 23.2% d⁻¹ in 1979 in Glen Lake. The long-term means (1977 to 1979) of each lake ranged from 3.89% d⁻¹ in the main basin of Red Chalk Lake to 22.31% d⁻¹ in Glen Lake with a mean of 7.1% d⁻¹ (Table 92).

Using the three-year mean data (n = 16), community FR was strongly correlated (p < 0.01) with B_z and B_h; however, as community FR, B_z and B_h were all calculated

Table 91. Filtering rates (FR) of common herbivorous Crustacea in study lakes. Filtering rate calculated from average length of each taxon which was, in turn, calculated from average individual weight (w) of each taxon using indicated relationships.

Species	Weight (μg animal ⁻¹)	Length Code ¹	Length (mm)	Filtering Rate Code ²	Filtering Rate (mL animal ⁻¹ d ⁻¹)	FR/W (mL ⁻¹ μg ⁻¹ d ⁻¹)
<i>Bosmina longirostris</i>	0.54	1	0.29	1	0.35	0.65
<i>Ceriodaphnia</i> sp.	0.50	2	0.44	2	1.71	3.42
<i>Chydorus</i> sp.	0.90	3	0.37	1	0.60	0.67
<i>Daphnia ambigua</i>	3.00	2	0.84	3	5.62	1.87
<i>D. catawba</i>	4.00	2	0.94	3	7.60	1.90
<i>D. dubia</i>	4.50	2	0.96	3	5.08	1.13
<i>D. galeata mendotae</i>	8.76	2	1.22	3	15.29	1.75
<i>D. longiremis</i>	5.00	2	1.00	3	8.97	1.79
<i>D. pulex</i>	8.09	2	1.18	4	9.89	1.22
<i>D. retrocurva</i>	3.14	2	0.85	3	5.81	1.85
<i>D. rosea</i>	8.00	2	1.18	5	24.09	3.01
<i>Diaphanosoma</i> sp.	1.89	2	0.71	6	6.48	3.43
<i>Eubosmina longispina</i>	0.54	1	0.29	1	0.35	0.65
<i>E. tubicen</i>	0.40	1	0.26	1	0.28	0.70
<i>H. gibberum</i>	10.90	4	1.19	7	14.68	1.35
nauplius	0.11	5	0.23	8	0.53	4.82
calanoid copepodid	0.87	6	0.47	8	1.03	1.18
<i>Diaptomus minutus</i>	2.24	6	0.69	8	1.47	0.66
<i>D. oregonensis</i>	3.55	6	0.84	8	1.77	0.50

¹ Length-weight relationships (from Sprules unpub. data)

1. $W = 266 L^{3.13}$ 4. $W = 70 L^{2.56}$

2. $W = 50 L^{2.84}$ 5. $W = 42 L^{2.48}$

3. $W = 297 L^{3.48}$ 6. $W = 55 L^{2.46}$

² Filtering rates from P. Chow (unpub. studies)

1. $\log FR = 2.174 \log L + 0.713$ 5. $\log FR = 3.461 \log L + 1.133$

2. $\log FR = 3.523 \log L + 1.489$ 6. $\log FR = 2.421 \log L + 1.172$

3. $\log FR = 2.680 \log L + 0.953$ 7. $\log FR = 1.955 \log L + 1.019$

4. $\log FR = 1.998 \log L + 0.852$ 8. $\log FR = 0.94 \log L + 0.32$

from zooplankton densities this simply represents a self correlation.

Community FR was not correlated ($p > 0.10$) with 1977 to 1979 means of ice-free season values of Secchi transparency ($r = -0.08$), total chlorophyll a ($r = 0.17$), phytoplankton biomass ($r = 0.33$) or diatom corrected phytoplankton biomass ($r = 0.17$). However, as for B_z and B_h , community FR was correlated with TP ($r = 0.76$, $p < 0.01$). Given the strong B_z - TP correlation ($r = 0.80$, $p < 0.01$), this was not surprising.

The stepwise multiple linear regressions with Secchi depth, chlorophyll a , phytoplankton biomass and diatom corrected phytoplankton biomass (three-year averages of ice-free season and summer stratified season) as dependent variables were rerun with three-year average community FR ($n = 16$) added to the array of independent variables. Inclusion of FR did not effect any changes in resulting regression equations.

The eight regression equations contained 25 independent variables (as previously) and in only 1 of the 25 cases did community FR explain a significant ($p < 0.10$) portion of the residual variance in the dependent variable. Clearly, this could have been a chance occurrence. Hence, even though FR/W was not constant for all herbivores, explicit consideration of calculated filtering rates did not improve predictability of indicators of trophic state.

Table 92. Calculated community filtering rate (% d⁻¹) by herbivorous crustacean zooplankton in study lakes.

Lake	Filtering Rate ¹			
	1977	1978	1979	mean
Blue Chalk	5.05	6.02	8.45	6.51
Chub	4.50	4.12	5.46	4.69
Dickie	10.85	3.12	4.57	6.18
Harp	3.72	4.00	5.05	4.26
Jerry	2.70	4.17	7.61	4.83
Red Chalk Main	3.18	4.19	4.30	3.89
Red Chalk East	4.22	8.59	6.86	6.56
Basshaunt	— ²	4.73	6.98	5.86
Bigwind	4.84	6.74	6.81	6.13
Buck	1.96	4.19	5.67	3.94
Crosson	5.59	9.85	9.10	8.18
Glen	— ²	21.41	23.20	22.31
Gullfeather	4.20	14.42	13.91	10.84
Little Clear	8.17	10.14	8.69	9.00
Solitaire	3.59	4.98	7.20	5.26
Walker	2.36	4.05	8.74	5.05

¹ Filtering rate calculated by summing the products of average filtering rate of each species (mL L⁻¹ d⁻¹ animal⁻¹) with its associated average density (animals L⁻¹). Individual filtering rates in Table 89.

² no data

4. WATER CLARITY

Water clarity in lakes is governed by the amount (and size distribution) of particulate material, the amount and colour of dissolved organic matter and by the inherent absorption of light by pure water (Brezonik 1978). The particulate matter includes bacteria, pigment-containing phytoplankton, zooplankton, non-living autochthonous

material (e.g., dead plant and animal matter), and organic and inorganic allochthonous materials (e.g., leaf litter particles, silt, clay, etc.). The use of water clarity as an indicator of trophic status is based on the assumption that transparency is principally determined by the quantity of pigment-containing phytoplankton. This belief has been supported by the observation that Secchi disc depth and chlorophyll a concentration are highly correlated in many lakes and sets of lakes (Edmondson 1972, Bachman and Jones 1974, Carlson 1977). In Ontario, Dillon and Rigler (1975) demonstrated that a hyperbolic relationship existed between mean chlorophyll a concentration and Secchi disc depth during summer stratification. The relationship can be written as:

$$SD_{SS} = 5.21 / (Chl\ a_{SS})^{0.41} \quad (47)$$

where SD_{SS} is the mean Secchi disc depth (m) during summer stratification. However, it was observed that the predictive capability of this relationship for dystrophic lakes (those with high concentrations of dissolved organic matter) was poor. Attempts to construct a model to predict Secchi disc depth (during summer stratification and during the ice-free period) that considers dissolved organic matter in addition to particulate matter were therefore made.

Mean Secchi disc depth data collected for both the "A" and "B" lakes in each of the four years are summarized in Table 93. The four-year averages for the summer stratification and ice-free periods are summarized in Table 94. Variation between years was not great, with the average coefficient of variation being 15% and 13% for the two time periods. The range in mean Secchi disc depth was not large (2.7 m - 7.5 m). Models for prediction of the mean Secchi disc depth during the two time periods were derived using stepwise multiple regression. The independent variables utilized included all of those available for the chlorophyll a and phytoplankton biomass models (Table 75), plus the phytoplankton parameters themselves (chlorophyll a , total phytoplankton biomass, corrected phytoplankton biomass).

For both the ice-free and summer stratification periods, Secchi disc depth was most strongly correlated (Table 95) with mean annual dissolved organic carbon (DOC_{AN}). In each case, this variable explained $\approx 88\%$ of the variance in Secchi disc depth. Secchi disc depth was also strongly correlated with all nitrogen fractions that included organic nitrogen ($TONEP$, $TNEP$, $TKNEP$), with chlorophyll a (CHL_{SS}), total phosphorus ($TPEP$ and/or $TPEM$) and water colour (COL_{IF}). DOC was very strongly correlated with water colour, each being a quantitative index of the degree of dystrophy of the lake. DOC was also strongly correlated with the nitrogen and phosphorus parameters, since the dissolved organic matter necessarily contains N and P as well as C.

Stepwise regression models, in both cases, selected TP (at spring overturn) after DOC (annual) as the second significant independent variable (Table 96). TP_{SO} has been previously used in many models to predict biomass of phytoplankton in the subsequent summer. However, none of the phytoplankton parameters themselves were selected as the second entry. This may be explained by

Table 93. Mean annual Secchi disc depth (m) for the summer stratification (SD_{SS}) and ice-free periods (SD_{IF}) for the years 1976 to 1979.

Lake	SD _{SS}				SD _{IF}			
	1976	1977	1978	1979	1976	1977	1978	1979
Blue Chalk	7.23	7.16	5.69	6.45	7.16	7.24	5.77	6.53
Chub	3.97	4.09	3.11	2.46	3.85	3.90	3.01	2.45
Dickie	3.12	3.18	3.03	2.28	2.96	2.98	2.96	2.30
Harp	4.41	4.33	3.40	3.56	4.46	4.32	3.37	3.54
Jerry	4.69	4.43	3.60	3.44	4.69	4.30	3.50	3.38
Red Chalk Main	6.76	6.50	5.50	6.21	6.65	6.46	5.42	5.96
Red Chalk East	— ¹	5.28	4.87	4.74	— ¹	5.22	4.80	4.68
Basshaunt	5.64	5.90	4.82	4.06	5.33	5.47	4.54	4.13
Bigwind	6.87	4.56	3.60	4.78	6.44	4.78	3.96	4.87
Buck	7.21	7.88	5.79	5.64	6.70	6.98	5.41	5.18
Crosson	3.80	4.53	3.62	3.22	3.74	4.10	3.37	3.00
Glen	4.48	— ¹	4.74	5.78	4.07	4.60	4.73	5.19
Gullfeather	2.80	3.22	2.38	2.46	2.73	2.90	2.18	2.33
Little Clear	5.91	6.27	4.85	4.74	5.62	5.46	4.92	4.74
Solitaire	7.94	8.30	7.00	6.82	7.57	8.42	7.05	7.55
Walker	5.47	6.70	5.35	4.98	5.24	6.62	5.05	5.29

¹ no data

the fact that TP may be a surrogate for all particulate material including live phytoplankton, living material other than phytoplankton, and non-living material. If this hypothesis were true, it would be expected that TN and TON would also be highly correlated (after DOC) with

Table 94. Mean Secchi disc depth (m) in the summer stratification (SD_{SS}) and ice-free periods (SD_{IF}). Coefficient of variation (c.v.) between years expressed as %. Values are calculated as the means of 4 annual means for the years 1976 to 1980.

Lake	SD _{SS}	(c.v.)	SD _{IF}	(c.v.)
Blue Chalk	6.63	(11)	6.67	(10)
Chub	3.41	(23)	3.30	(21)
Dickie	2.90	(14)	2.80	(12)
Harp	3.92	(13)	3.92	(14)
Jerry	4.04	(15)	3.97	(16)
Red Chalk Main	6.24	(9)	6.12	(9)
Red Chalk East	4.96	(6)	4.90	(6)
Basshaunt	5.10	(16)	4.86	(13)
Bigwind	4.95	(28)	5.01	(21)
Buck	6.63	(17)	6.07	(15)
Crosson	3.79	(15)	3.55	(13)
Glen	5.00	(14)	4.65	(10)
Gullfeather	2.72	(14)	2.53	(13)
Little Clear	5.44	(14)	5.18	(8)
Solitaire	7.51	(10)	7.65	(7)
Walker	5.63	(13)	5.55	(13)
	mean (15)		mean (13)	

Secchi disc depth; this was, in fact, observed. Mean annual chlorophyll *a* concentration was selected as the third parameter in the ice-free period model, while maximum depth and mean depth were added to the model for the summer stratification period. Since the latter independent variables improved the predictive capabilities of the two models very little, models were constructed with only DOC_{AN} and TP_{SO} included (Table 97):

$$SD_{SS} = -1.03 \text{ DOC}_{AN} - 0.046 \text{ TP}_{SO} + 10.15 \quad (48)$$

$$SD_{IF} = -1.03 \text{ DOC}_{AN} - 0.061 \text{ TP}_{SO} + 10.14 \quad (49)$$

Table 95. Correlation between mean Secchi disc depth during summer stratification (SS) and the ice-free periods (IF). Independent variables and their units are listed in Table 75.

variable	SD _{SS}		variable	SD _{IF}	
	r ¹	F ²		r ¹	F ²
DOC _{AN}	-0.940	106.5	DOC _{AN}	-0.938	102.7
TON _{EP}	-0.835	32.3	TON _{EP}	-0.855	37.9
TN _{EP}	-0.835	32.3	TN _{EP}	-0.854	37.7
TKN _{EP}	-0.824	29.6	TKN _{EP}	-0.845	34.8
CHL _{SS}	-0.808	26.3	TP _{EP}	-0.818	28.4
TP _{EP}	-0.806	26.0	TP _{EM}	-0.805	25.8
COL _{IF}	-0.793	23.7	CHL _{SS}	-0.794	23.9

¹ correlation coefficient

² partial F-value

Table 96. Stepwise regression models for the prediction of Secchi disc depth in the summer stratification period (SD_{SS}) and the ice-free period (SD_{IF}). Independent parameters and their units are listed in Table 75.

	Step 1	Step 2	Step 3	Step 4	Constant	p	SE
SD _{SS} ¹	-0.99 DOC _{AN}	-0.075 TP _{SO}	-0.094 Z _{MAX}	0.202	10.81	<0.001	0.344
SD _{IF} ²	-1.26 DOC _{AN}	-0.065 TP _{SO}	-0.039 CHL _{AN}	—	10.27	<0.001	0.384

¹ r² = 0.956; r₁² = 0.884; r₂² = 0.025; r₃² = 0.019; r₄² = 0.028

² r² = 0.941; r₁² = 0.880; r₂² = 0.042; r₃² = 0.019

Table 97. Stepwise regression models truncated after second step for the prediction of average Secchi disc depth in the summer stratification period (SD_{SS}) and the ice-free period (SD_{IF}). Independent parameters and their units are listed in Table 75.

	Step 1	Step 2	Constant	p	SE
SD _{SS} ¹	-1.03 DOC _{AN}	-0.046 TP _{SO}	10.15	<0.001	0.454
SD _{IF} ²	-1.03 DOC _{AN}	-0.061 TP _{SO}	10.14	<0.001	0.425

¹ $r^2 = 0.909$; $r_1^2 = 0.884$; $r_2^2 = 0.025$

² $r^2 = 0.922$; $r_1^2 = 0.880$; $r_2^2 = 0.042$

These models are recommended for predicting water clarity in lakes for which the range in independent variables falls within those observed in the study lakes. In cases where phytoplankton biomass or chlorophyll *a* concentration is much higher ($>1 \text{ mg L}^{-1}$, or $>5 \mu\text{g L}^{-1}$, respectively), these models will overestimate water clarity.

E. AREAL HYPOLIMNETIC OXYGEN DEFICIT (AHOD)

The rate of loss of oxygen from the hypolimnion (or more properly, from the tropholytic zone – the zone where decomposition rate exceeds primary production rate) in lakes has been used for many years as a measure of trophic status (Hutchinson 1938). Although the loss of oxygen is a result of a combination of processes – decomposition of autochthonous matter that was produced in the euphotic zone then settled into the hypolimnion or tropholytic zone, decomposition of allochthonous organic matter (dissolved, colloidal and particulate), decomposition of particulate organic matter in the surficial sediments – the AHOD rate of a lake is an effective state variable dependent upon the overall trophic status of the lake.

Because hypolimnetic oxygen concentration determines, to a significant extent, the suitability of a lake for some fish populations, it would be useful to predict the AHOD, and in combination with lake morphometry, the oxygen concentration as a function of depth, in any lake. Only preliminary attempts to construct a model for the prediction of AHOD based on the Lakeshore Capacity Study lakes have been completed. A more detailed study of AHOD will be published separately.

In recent years, three models have been proposed for the prediction of AHOD. Charlton's (1980) model has been tested primarily for the Laurentian Great Lakes. Cornett and Rigler's (1979) and Welch and Perkins' (1979) models were developed using information collected for small lakes, more similar to those investigated in the Lakeshore Capacity Study. In light of this, the predictive capability of the Cornett and Welch models using AHOD's calculated for the "A" study lakes were compared.

Annual AHOD measured for each of the "A" lakes is shown in Table 98. Variation between lakes was small (less than two-fold), with variation between years as great or greater. Details of the oxygen regimes of the study lakes are provided by Reid et al. (1983a).

Table 98. Areal hypolimnetic oxygen depletion rate (AHOD; $\text{mg O}_2 \text{ m}^{-2} \text{ d}^{-1}$) for the "A" lakes.

Lake	AHOD				mean
	1976	1977	1978	1979	
Blue Chalk	214	226	185	161	197
Chub	349	342	256	181	282
Dickie	305	259	261	271	274
Harp	429	501	343	240	378
Jerry	317	495	318	265	349
Red Chalk	307	481	316	186	323

The Welch model is:

$$\log \text{AHOD} = 1.58 + 0.37 \log (L_p \cdot \tau_w) \quad (50)$$

while the Cornett model can be expressed as:

$$\text{AHOD} = 0.5 S - 277 + 150 \ln \bar{z}_H \quad (51)$$

where *S* is the TP retained in the lake each year ($\text{mg m}^{-2} \text{ yr}^{-1}$), and $\bar{z}_H$ is the mean thickness of the hypolimnion.¹

Total phosphorus input (Table 99) was calculated as the sum of the natural (measured) input plus, for those lakes with shoreline development, the inferred anthropogenic input (see Section III.B.10). The total phosphorus retained (*S*) was calculated as the difference between the total input and that lost by outflow (Section III.B.9). Comparison of the Welch and Cornett models indicated that although the predictions based on the Welch model were closer to the observed AHOD's than those based on the Cornett model, the correlation between predicted and observed values was much greater with the Cornett model. This fact suggested that re-calibration of the model coefficients would yield improved predictive capability. When this was done, the resulting Cornett-type model:

$$\text{AHOD} = 1.86 S - 52 + 99.1 \ln \bar{z}_H \quad (52)$$

provided by far the closest agreement between measured and predicted AHOD's for the study lakes, with the mean square error reduced by 99% from the original Cornett model and 83% from the original Welch model.

The model represented by equation 52 must be tested on an independent data set before it can be considered to be a useful predictive model.

Table 99. Estimated TP load (sum of the measured natural load, Tables 53-58, and the estimated anthropogenic load – section III.B.10; in $\text{mg m}^{-2} \text{ yr}^{-1}$) and the mean thickness of the hypolimnion ($\bar{z}_H$, m).

Lake	L _p				mean	$\bar{z}_H$
	1976-77	1977-78	1978-79	1979-80		
Blue Chalk	79.9	61.7	53.8	47.9	60.8	6.2
Chub	135.9	121.1	109.7	129.3	124.0	6.8
Dickie	197.1	176.0	166.1	178.8	179.5	1.5
Harp	143.7	115.4	125.1	112.2	124.1	10.0
Jerry	224.2	158.7	204.2	163.1	187.5	10.0
Red Chalk	101.6	98.8	89.3	87.2	94.2	11.2

¹ A third term, dependent upon hypolimnetic temperature, has been excluded for this analysis since its contribution to the model is very small.

F. STEPWISE PROCEDURE FOR CALCULATING THE EFFECT OF SHORELINE DEVELOPMENT ON THE TROPHIC STATUS OF LAKES

The conceptual model that formed the basis for the research carried out by the Trophic Status Component of the Lakeshore Capacity Study was illustrated in Figure 1. In sections III and IV, the outcome of this research, a series of mathematical relationships (models) describing the linkages in the conceptual model, was presented. This section provides a stepwise procedure describing use of the model for determining the effects of shoreline development on variables which describe the trophic status of a lake including the concentration of TP; algal abundance expressed as phytoplankton biomass and chlorophyll *a* concentration; water clarity as expressed by Secchi disc depth; and hypolimnetic oxygen conditions as expressed by the areal hypolimnetic oxygen deficit (AHOD).

The model can also be used to determine the capacity of a specific lake for development if an acceptable trophic status is specified.

The model described in this report was developed on the basis of data collected for a series of small, oligotrophic and mesotrophic lakes. Extrapolation of this model to lakes that are very different (e.g. extremely large, or eutrophic) is not warranted without further research.

Most of the information needed to utilize the model can be obtained from existing data bases including aerial photographs, topographic and geological maps, government reports, and other published literature. In some steps of the model, several options exist for performing the required calculations. Those options utilizing observed data rather than predicted information should be used wherever possible.

Step 1: Assemble basic information.

Obtain from existing data bases (or measure if necessary) lake surface area (A_o , m^2), lake volume (V , m^3) and the area of the subwatershed of each inlet to the lake (A_d , m^2). This information will be used in several steps of the model. Other basic information, required at certain steps or for certain calculation options, will be described as it is needed.

Step 2: Calculate anthropogenic TP input (J_A).

Obtain information on the total number of units (n) (including any existing ones) to be situated on the shoreline of the lake (defined here as within 300 m of the lake or any inflowing stream to the lake). Additionally, obtain information on any proposed large-scale alterations in land use (e.g. clearcutting of forest, dredging or filling of lowland areas) which may affect the TP input from the terrestrial watershed.

Using the appropriate methodology outlined in Downing (1983), calculate the number of capita years yr^{-1} that will be spent at the lake (usage).

Assume a value of 0.8×10^6 mg capita $^{-1}$ yr^{-1} as the anthropogenic TP supply to the sewage disposal facility. From information on the type of existing and proposed sewage systems, determine the retention capacity (R_s) of each system as follows:

$R_s = 0$ for conventional septic tank/tile field systems, leaching pits and aerobic systems. If there is evidence that the septic system is in good order and information on the soil properties is available, then calculate a value for R_s from Table 29.

$R_s = 1$ for holding tanks if waste disposal is outside of the watershed.

Calculate the anthropogenic TP supply to the lake (J_A , mg yr^{-1}) as follows:

$$J_A = 0.8 \times 10^6 \cdot \text{usage} \cdot (1 - R_s)$$

This calculation assumes that all anthropogenic TP reaches the lake the same year that it is supplied to the sewage disposal system (i.e. lag time = 0).

Step 3: Calculate TP input from precipitation (J_{PR}).

Using the estimated TP deposition of 37.3 mg m^{-2} yr^{-1} and the lake area A_o (m^2), calculate the annual TP input (J_{PR} , mg yr^{-1}) to the lake from precipitation as follows:

$$J_{PR} = 37.3 \cdot A_o$$

Step 4: Calculate natural TP input from terrestrial watershed (J_N).

The method chosen to determine TP input from the watershed (J_N) is dependent upon the detail of information on watershed characteristics available. The following calculations must be performed for each inlet to the lake. If extensive data on the hydrological and geological characteristics of the watershed are available (Table 70), use the following relationship to predict TP export (E in mg m^{-2} yr^{-1}) from the watershed:

$$E = 0.38V27 + 0.012V2 - 0.33V8 - 0.29V28 - 23.85V43 + 1.43$$

If no hydrological information is available, calculate E as follows:

$$E = 0.46V27 - 0.27V8 - 0.28V28 + 0.053V29 + 5.99$$

If the geologic information is not sufficiently detailed to allow use of the above equation, TP export may be estimated knowing basic characteristics of land use and bedrock geology as outlined in Table 26.

Using the calculated values of E and the drainage basin area (A_d , m^2), calculate the TP input to the lake (J_N in mg yr^{-1}) from the n watersheds as follows:

$$J_N = \sum_{i=1}^n E_i \cdot A_{di}$$

Take lakes upstream into account by starting the calculations at the headwater lake of the chain. The fraction of TP supplied to the lake that is retained by the lake is calculated as outlined in Step 6.

Step 5: Calculate hydrological characteristics of lake (Q , q_s , τ_w).

The primary hydrological characteristic is lake outlet discharge (Q in m^3 yr^{-1}) and the method of estimation depends upon the amount and type of available information. If daily precipitation depth, daily meteorological data, and the physical and geological characteristics of the watershed are available, use the modified TVA model to calculate runoff. Instructions for the use of this model are

too lengthy to present in this summary. If this detailed information is not available and estimates of the long-term mean Q are adequate, use Pentland (1968) or the Hydrological Atlas of Canada (1978) to estimate runoff (r in m yr^{-1}). Annual lake outflow volume may be calculated as follows:

$$Q = r \cdot A_d$$

Calculate areal water loading rate (q_s in m yr^{-1}) as follows:

$$q_s = Q/A_o$$

and the theoretical water replenishment rate (τ_w in yr^{-1}) as follows:

$$\tau_w = Q/V$$

Step 6: Calculate retention of TP by lake (R_p).

Use existing information (or direct measurements) to determine the oxygen levels of the hypolimnion. If the hypolimnion experiences periods of anoxic conditions, the retention of TP by the lake is calculated as follows:

$$R_p = 7.2/(7.2 + q_s)$$

Otherwise, calculate the TP retention as follows:

$$R_p = 12.4/(12.4 + q_s)$$

As it is not currently possible to reliably predict the oxygen levels in the hypolimnion, measurements are necessary.

Step 7: Calculate the concentration of TP in the lake (TP_{IF}).

Using estimates of J_A , J_{PR} and J_N derived in steps 2-4, calculate the total load of TP to the lake (L in $\text{mg m}^{-2} \text{yr}^{-1}$) as follows:

$$L = \frac{J_A + J_{PR} + J_N}{A_o}$$

Calculate the mean ice-free concentration of TP in the lake as follows:

$$[TP]_{IF} = L(1 - R_p)/0.956 q_s$$

If information is available on the concentration of TP in the lake at other time periods, calculate the mean ice-free period $[TP]$ using the relations in Table 48.

Step 8: Calculate the response time of the lake (RT).

The response time of the lake (RT in years) gives an indication of the time required for the concentration of TP in the lake to respond to an altered TP supply. Response time can be expressed in terms of the half-life ($t_{1/2}$), the time required for the TP concentration in the lake to change half-way from its original steady-state level to its final steady-state level. It is suggested that 3 times the $t_{1/2}$ (or 87.5% of the time required to reach the final steady-state TP concentration) be used as an indicator of response time. Calculate response time as follows:

$$RT = 3t_{1/2} = 2.07/(1/\tau_w + v/\bar{z})$$

where $v = 12.4$ for lakes with oxic hypolimnia

and $v = 7.2$ for lakes with anoxic hypolimnia

Step 9: Calculate algal standing stock (chl a_{IF} , PB_{IF} , CPB_{IF}).

Three estimates of mean ice-free algal standing stock may

be calculated (chl a_{IF} – chlorophyll a ; PB_{IF} – phytoplankton biomass; CPB_{IF} – phytoplankton biomass excluding diatoms). The choice of equations used to estimate algal standing stock depends upon the type of available information. Where two equations are given, use the first one if the information is available. The independent variables used in the equations are listed in Table 75.

Calculate chlorophyll a concentrations (chl a , mg m^{-3}) using one of the two following equations:

$$(chl a)_{IF} = 0.83 TP_{EP} + 0.12 (N/P)_{EP} - 0.018 TIN_{EP} - 0.0076 TON_{IC} - 5.56$$

$$(chl a)_{IF} = 0.48 TP_{EP} + 0.060 (N/P)_{EP} - 3.14$$

If no data on N/P ratios in the epilimnion are available, use the following relation based on N/P ratios at fall overturn:

$$(N/P)_{EP} = 0.43 (N/P)_{FO} + 19.4$$

Calculate phytoplankton biomass (PB_{IF} in mg m^{-3}) using the following equation:

$$PB_{IF} = 0.068 TP_{EM} + 0.0044$$

Calculate phytoplankton biomass excluding diatoms (CPB_{IF}) using one of the following two equations:

$$CPB_{IF} = 0.25 TP_{EP} + 0.039 (N/P)_{EP} - 0.0053 TN_{EP} + 0.080 pH_{EP} - 2.10$$

$$CPB_{IF} = 0.098 TP_{EP} + 0.012 (N/P)_{EP} - 0.84$$

Step 10: Calculate water clarity (SDIF).

Based on the amount of data available, calculate the water clarity as expressed by Secchi depth (SD_{IF}) using one (use the first equation if the data is available) of the following equations:

$$SD_{IF} = -1.26 DOC_{AN} - 0.065 TP_{SO} - 0.39 (chl a)_{AN} + 10.27$$

$$SD_{IF} = -1.03 DOC_{AN} - 0.061 TP_{SO} + 10.14$$

Measurements of DOC will have to be taken as no method currently exists for predicting DOC levels. If no data are available for TP_{SO} , it may be related to TP_{IF} using the following equation:

$$TP_{SO} = 1.18 TP_{IF} - 1.91$$

Step 11: Calculate the areal hypolimnetic oxygen deficit (AHOD).

Calculate the areal hypolimnetic oxygen deficit (AHOD in $\text{mg O}_2 \text{m}^{-2} \text{d}^{-1}$) from the following equation:

$$AHOD = 1.86 S - 52 + 99.1 \ln \bar{z}_H$$

$\bar{z}_H$ (mean depth of the hypolimnion) may be estimated if the lake morphometry and the depth of the thermocline (D_T) are known. D_T may be calculated as follows (Ragotzkie 1978):

$$D_T = 4\sqrt{F}$$

where F = fetch of lake (km).

$$S = L \cdot R_p$$

As the equation predicting AHOD has not been tested on an independent data set as yet, it should be applied with caution.

If no data on $\bar{z}_H$ is available, calculate AHOD as follows:

$$\log AHOD = 1.58 + 0.37 \log (L \cdot \tau_w)$$

V. ADDITIONAL WORK

In addition to the components of the trophic status model described in the previous sections, preliminary work was carried out on two related subjects.

Zooplankton biomass in the study lakes was measured to investigate the possibility of improving nutrient-phytoplankton relationships (see Section IV.D.3). However, the potential for predicting zooplankton biomass was also explored since the capability of predicting zooplankton biomass may be an important intermediate step in predicting fish biomass. This is discussed in the subsequent sub-section.

In the trophic status modelling that was undertaken in this study, total phosphorus was the only form of phosphorus considered. Reasons for the emphasis on this fraction of phosphorus have been discussed by Dillon and Reid (1981). However, the fraction of TP that is biologically available (BAP) is generally unknown. If appropriate methods of measurement of BAP existed, better trophic status models could almost certainly be produced.

The first steps towards development of a suitable method of measuring BAP were made as part of this study. A bioassay based on phosphorus uptake rate measurements was successfully developed. The method is described in detail by Dillon and Reid (1981) and Mierle (1983). A preliminary investigation of the availability of the phosphorus in some of the major inputs to the study lakes demonstrated that the TP input via atmospheric deposition was totally available to biological organisms, while only a fraction (10-60%) of that in several of the streams was available. These results indicate that the TP from different sources may not be of equal importance in determining the trophic status of lakes. These studies will be described in detail in a separate report.

A. EMPIRICAL PHOSPHORUS - ZOOPLANKTON BIOMASS RELATIONSHIPS

In recent years limnologists have developed a large number of empirical regression models which can be used to predict phytoplankton biomass (PB) or chlorophyll *a* (chlor *a*). Fisheries biologists similarly have developed empirical relationships for use in predicting fish community yield or biomass from such independent variables as total dissolved solids (Rawson 1951, 1960) mean depth (Rawson 1952, Prepas 1983), the ratio of total dissolved solids to mean depth (Ryder et al. 1974), phytoplankton productivity or biomass (Oglesby 1977, Biro and Vörös 1982), macrobenthic biomass (Matuszek 1978) and, finally, TP levels (Hanson and Leggett 1982).

Since the pioneering work of Rawson (1953) and Northcote and Larkin (1956) there has been little effort directed at developing such models for zooplankton biomass. Given the intermediate tropho-dynamic position of zooplankton between phytoplankton and fish, such models may be of interest. Surveys of existing literature from lakes in various regions have indicated that over large gradients in nutrient status good correlations may exist between phytoplankton biomass or productivity and zooplankton biomass or productivity (Brylinsky and Mann 1973, Mackarewicz and Likens 1979, McCauley and Kalff 1981). However no relationships have been constructed (since Northcote and Larkin 1956) using data collected with that purpose in mind from a single geographic region, for example, for Canadian Shield Lakes. Further, the usefulness of nutrients, particularly TP, as predictors of zooplankton biomass has only recently been investigated (Hanson and Peters 1984).

Earlier in this report (Section IV.D), it was shown that TP was a good predictor of PB in the study lakes even though the range in TP levels was small. In this section, the usefulness of readily obtainable morphometric, chemical and phytoplanktonic information for predicting average total crustacean zooplankton biomass (B_z) in the LCS lakes is assessed. As zooplankton may be important in the diet of Ontario fish (Martin 1952, Keast 1973), it is anticipated that such predictions would be of interest to fisheries biologists.

Zooplankton were enumerated in whole-lake volume-weight composite samples collected roughly on a weekly and monthly basis from the "A" and "B" lakes, respectively, as described by Scheider et al. (1983). Biomass in each sample (mg m^{-3} dry weight) was calculated from the enumerated samples, using average species dry weights. The reported values of B_z are time-weighted arithmetic means of biomass for the ice-free seasons of 1977, 1978 and 1979. Three-year means are used in an attempt to reduce uncertainty associated with low total numbers of samples in the "B" lakes (Table 88) and to eliminate the effects of lags in population responses which may confound shorter-term data.

Among the "A" lakes, B_z varied by less than two-fold from 37.1 mg m^{-3} in the main basin of Red Chalk Lake to 58.2 mg m^{-3} in the small eastern basin. B_z ranged from 33 to 104 mg m^{-3} among the "B" lakes (Table 88). These values are at the low end of the range of values typically observed in oligotrophic Shield lakes (Schindler and Novén 1971).

Product moment correlation coefficients of B_z with 16 limnological variables are given in Table 100. Correla-

tions were poor ($p > 0.10$) for Secchi depth, chlor *a* and PB, for the chemical parameters, pH, colour, DOC and TIN/TP and for two morphometric parameters, A_o and V. Significant positive correlations were observed for TP, TON, TIN and TN, and significant negative correlations with TN/TP, $\bar{z}$ and z_{max} . While not indicated in the table, correlations with annual and summer stratified averages were of similar sign and magnitude as those with ice-free season averages.

Table 100. Correlation coefficients (*r*) between B_z and a variety of limnological variables. All chemical and biological data were three year means. Single and double asterisks indicate $P < 0.05$ and $P < 0.01$, respectively.

Parameter	<i>r</i>
TP	0.80**
TON	0.52*
TIN	0.70**
TN	0.75**
TN/TP	-0.54*
TIN/TP	-0.05
Secchi	-0.30
Chlor <i>a</i>	0.28
PB	0.38
pH	0.29
Colour	0.40
DOC	0.30
A_o	-0.26
$\bar{z}$	-0.57*
z_m	-0.65**
V	-0.42*

There are three possible reasons why the correlation of B_z with nitrogen and particularly with phosphorus levels may have been stronger than that with expressions of phytoplankton standing stock. First, herbivorous zooplankton feed on bacteria and tripton (non-living suspended material) in addition to phytoplankton. Gliwicz (1969) in fact estimated that in oligotrophic lakes in Poland, phytoplankton form a maximum of 50% of the diet of planktonic herbivores. Measured TP is probably a better measure of total grazable material in the study lakes than is PB or chlor *a*.

Secondly, the impact of herbivores on phytoplankton is negative at times, but as recyclers of essential nutrients zooplankton may also have a net positive impact on phytoplankton at other times (Lehman 1980, Redfield 1980). The poor correlations of B_z with PB may, therefore, be a reflection of temporal changes in the kind of phytoplankton-zooplankton interaction. Finally, the fraction of PB which is actually grazable may be overestimated by BP and may vary between lakes, as zooplankton may graze selectively upon the available phytoplankton species (McNaught et al. 1980).

All variables listed in Table 1 were used as the independent variable matrix in a stepwise multiple linear regression analysis using B_z as the dependent variable. With the F ratio to add or delete variables set at 95%, the resulting model had TP as the single independent variable:

$$B_z = 14.32 + 4.46 \text{ TP} \quad (r^2 = 0.63, F = 24.1, P < 0.01)$$

Given the successful application of TP as a predictor of community standing stocks at other trophic levels, it is not surprising that it is the best univariate predictor of zooplankton biomass in the study lakes.

McCauley and Kalff (1981) assembled phytoplankton biomass and zooplankton biomass data from 17 lakes in various parts of the world in an effort to construct a PB- B_z relationship. Over their wide range in PB (0.4 to 50 mg L⁻¹ wet weight), they found a good relationship existed between the parameters (both in $\mu\text{g L}^{-1}$ wet weight) as below:

$$\log_{10} B_z = 0.719 \log_{10} \text{PB} + 1.015 \quad (r^2 = 0.72)$$

Because B_z and PB were weakly correlated in the study lakes (Table 100) it was not anticipated that this relationship could be used to predict B_z in the lakes. To test this, the data were converted to wet weights assuming a dry to wet weight ratio of 0.12, as did McCauley and Kalff, and predicted B_z using measured PB values. As expected, predicted and measured B_z values were very poorly correlated ($r = 0.39$, $p > 0.10$, $n = 16$), and the slope of an equation of predicted vs. observed B_z was only 0.2, much less than the ideal of 1.0. This indicates that the McCauley and Kalff relationship is not useful for predicting B_z in nutrient-poor Shield lakes.

McCauley and Kalff did not examine TP- B_z relationships to see if, as in the study lakes, these may have been stronger than PB- B_z relationships. The TP data were not available (McCauley, personal communication). An examination of two other data sets indicates that over a wider range in nutrient status than found in the study lakes, TP may indeed be a better predictor of B_z than PB. For seven lakes in the Kawartha region of Ontario with TP levels ranging from 16 to 55 mg m⁻³, TP and B_z were strongly correlated ($r = 0.92$, $p < 0.01$), but the correlation of PB with B_z was poor ($r = 0.46$, $p > 0.10$, Anon. 1976). Similarly, while Mills and Shrivane (1982) report a significant relationship between chlor *a* and B_z for 13 lakes in New York State ($r = 0.75$), correlation of B_z with TP levels calculated from presented data are stronger ($r = 0.84$).

VI. SUMMARY

The Lakeshore Capacity Study was initiated to provide quantitative models that would allow evaluation of the effects of shoreline development on lakes and their watersheds. This report summarizes the research carried out to estimate the effects of shoreline development on the trophic status of lakes. The results of these studies have been integrated with those of the other components to provide a sound technical basis for management of shoreline development.

The conceptual model linking shoreline development, morphometry, the hydrologic and TP budgets and trophic status of a lake was originally outlined by Dillon and Rigler (1975). The trophic status of a lake was described by several state variables including phytoplankton biomass, chlorophyll *a* (a surrogate measure of phytoplankton biomass), water clarity and the loss rate of oxygen from the profundal waters.

A critical untested assumption in the conceptual model was that shoreline development resulted in the input of TP to lakes via the sewage disposal systems. Furthermore, many linkages in the conceptual model were not sufficiently well developed to allow accurate predictions of trophic status with an acceptable degree of confidence. Further model development and testing were therefore necessary.

Two approaches were used in this study to address the question of whether shoreline development provided TP to lakes. In the first approach, the natural TP and hydrologic budgets were measured for six lakes ("A" lakes) exhibiting a large range in the potential significance of shoreline development to the TP budgets. Existing trophic status models were calibrated using the two undeveloped (no anthropogenic TP input) lakes of the set, then applied to the four developed lakes. Differences between measured TP concentrations in the developed lakes and TP concentrations predicted from the calibrated models were assumed to be attributable to anthropogenic TP inputs which, in turn, were quantitatively related to the extent of shoreline development.

In the second approach, the trophic status of nine selected lakes ("B" lakes) was compared before and after shoreline development. However, as few of these changes occurred over the course of the Lakeshore Capacity Study, this approach is not discussed at length in this report.

During the course of the study, relationships were developed, tested and refined to link individual components of the conceptual model. The models developed include those for prediction of the important components of the hydrology of lakes and watersheds, the yield or export of TP from the terrestrial watershed, the retention of TP in

the sediments of the lake, the concentration of TP in the lake, the phytoplankton biomass and chlorophyll *a* levels, the water clarity (as measured by Secchi disc), the zooplankton biomass, and the depletion rate of oxygen from the profundal waters. Because several of these relationships depended on the geology and geochemistry of the watersheds, an additional set of nine terrestrial watersheds ("export" watersheds) were chosen to extend the range in geological characteristics observed in the "A" and "B" lake watersheds.

Details of the methods employed in carrying out the studies are described in Scheider et al. (1983a). The study lakes are located in the County of Haliburton and the District Municipality of Muskoka in Ontario. Information on climate, lake morphometry, vegetation and geology are summarized in Scheider et al. (1983b), Nicolls et al. (1983), and Jeffries and Snyder (1983) respectively. All lakes exhibited pH, conductivity and major ion and nutrient concentrations typical for those observed in dilute, nutrient-poor Precambrian lakes. Average (1976-1980) concentrations of TP ranged from 5.7 mg m⁻³ to 12.8 mg m⁻³ for the "A" lakes and 6.4 mg m⁻³ to 20.7 mg m⁻³ for the "B" lakes. Using the trophic status classification scheme of Vollenweider and Kerekes (1980), all lakes but one could be classified as oligotrophic. A summary of the major findings regarding the individual components of the hydrologic and TP budgets of the "A" lakes follows.

Hydrologic budgets were calculated for each lake for four annual (June-May) periods and for the entire four-year period. In all cases except one, the budgets balanced to within $\pm 10\%$, thereby giving confidence in the individual supply and loss terms and in the use of data for construction of chemical mass balances and the calculation of water replenishment times for the lakes. A bias to negative balances (inputs < losses) was observed.

The major ion balances for the conservative ions Ca²⁺, Mg²⁺, Na⁺, K⁺, and Cl⁻ generally supported the contention that the hydrological budgets were accurately measured. The ion balances showed deficits in input, supporting the findings of the hydrologic budgets, and indicating a small systematic error in the mass balances.

The range in atmospheric deposition of TP was 21.8 mg m⁻² yr⁻¹ to 58.4 mg m⁻² yr⁻¹ with a mean (1976-1980) of 37.3 mg m⁻² yr⁻¹, indicating the need for long-term monitoring to obtain reliable estimates of deposition. A seasonal pattern in the deposition of TP was observed, peak values occurring in the summer months. Deposition rates of TP at all but one location in the study area were not significantly different, although significant differences were observed between collectors located on a

lake and those located under the forest canopy adjacent to the lake.

The mean (1976-1980) TP export (yield) for 30 watersheds ranged from $2.0 \text{ mg m}^{-2} \text{ yr}^{-1}$ to $28.5 \text{ mg m}^{-2} \text{ yr}^{-1}$. The grand mean TP export for metamorphic watersheds was $8.2 \text{ mg m}^{-2} \text{ yr}^{-1}$, somewhat higher than the mean ($5.5 \text{ mg m}^{-2} \text{ yr}^{-1}$) based on previous studies. Year-to-year variability in yield was not significant compared to site-to-site variability, although the monthly values of TP yield ranged over two orders of magnitude within each watershed with maximum values occurring during the spring snowmelt.

Based on a literature review, $0.8 \text{ kg TP capita}^{-1} \text{ yr}^{-1}$ was calculated as the average TP supply to sewage disposal systems in the area. Estimates of TP retention in the sewage disposal system were made for types of systems in use around the study lakes. Retention of TP in septic tank and tile field systems was assumed to be zero unless information on the condition of the system and soil type was available. A correction factor for TP retention in the soil may then be applied. Information on human usage and sewage disposal type obtained from the Land Use Component (Downing, pers. comm.) allowed estimates of potential anthropogenic TP input to the study lakes ranging from 0 kg yr^{-1} to 82.8 kg yr^{-1} .

The loss of TP to the lake sediments was calculated in three ways: from the measured TP mass balances (undeveloped lakes only); from sediment trap data; and from information on the concentration of TP in sediments and the sediment accumulation rate. A comparison of the latter two methods with the first method showed that sediment traps overestimated the TP sedimentation rate by 12 to 17 times, whereas good agreement was found between the mass balance and sediment chronology methods.

Minor components of the TP budget were investigated. For Harp Lake, net TP loss by fish harvest was estimated as 2% of the total TP supply, TP loss by emerging insects was estimated as being negligible and the TP supply by forest litterfall was estimated as 3% of the total TP input.

Based on the TP mass balance information, the relative importance of the various sources of TP to the lakes were assessed. The inflowing streams were of highly variable importance, providing from 0.4% to 69% of the natural TP input to the lakes. Precipitation input of TP was the single most important source of TP in four of the six "A" lakes, contributing 20%-74% of the total natural supply (1976-1980 mean values). Contributions of TP from the ungauged watershed areas were relatively small, averaging 15% of the total natural supply of TP to the lakes (range 8%-23%). The loss of TP to the sediments was always greater than the loss to the lake outlet, even if the potential anthropogenic inputs were not considered. Finally, the potential contributions of the anthropogenic sources of TP were calculated as 55%, 46%, 19%, 7%, 1% and 0% of the total TP supply for Harp, Dickie, Blue Chalk, Chub, Red Chalk and Jerry Lakes respectively. The year-to-year variability in the individual components of the TP budgets was great, indicating that measurement of TP mass balances for several years was necessary to develop and validate trophic status models.

The impacts of shoreline development on communities of phytoplankton, zooplankton and macrophytes were assessed on the "A" and "B" lakes by examining the correlations between parameters describing the state of the biological communities and indices of lakeshore development or by comparing the values of the parameters in sets of lakes of extremely different intensities of use. Of the 18 correlations run for phytoplankton, only one was significant and the development index explained only 28% of the variance in the parameter. Comparing phytoplankton community parameters for the three most developed and the four undeveloped lakes showed that there was no significant difference between the two sets of lakes. No detectable change was found in the composition, standing stock or nutritional status of macrophytes in eight lakes covering the full range of human use. Finally, comparison of ten zooplankton community parameters (1977-1979) in the three most developed and four undeveloped lakes showed no significant differences between lake sets.

A summary of the relationships developed, refined and tested to link individual components of the conceptual model are given below.

Existing models predicting the four principal hydrologic components (precipitation, runoff, evapotranspiration and evaporation) from readily obtainable hydrometeorological data were refined and tested on the "A" lakes and "export" watersheds. Precipitation at a given site could be predicted using data from nearby, existing meteorological stations. Of the five techniques investigated to predict runoff, the preferred method was the modified Tennessee Valley Authority (TVA) model because it allowed prediction of streamflow on a fine time scale (daily) for any given set of climatic conditions. After calibration on the "A" lake watersheds, the modified TVA model was tested on the "export" watersheds and gave good predictions of daily flow in all years except 1977-78. Both basin evapotranspiration and lake evaporation could be predicted based on the climatological models of Morton et al. (1980). For Harp Lake, estimates of annual lake evaporation using Morton's method agreed to within 2%-8% of those made using the more laborious energy balance method. Evapotranspiration rates for Harp and Jerry Lake watersheds estimated using Morton's method agreed to within 2% of those estimated using the water balance method over the 1976-1980 period.

Two models were developed to predict annual yield of TP from terrestrial watersheds. In the first model, 54% of the variance in the TP yield of 30 watersheds was related to geologic, land use and hydrologic characteristics of the watershed. In the second model, 46% of the variance in the TP yield was explained by geologic and land use characteristics alone.

The relationship given in Dillon et al. (1977) successfully predicted the annual retention of TP from the areal water load when tested against estimates derived from the TP mass balances in Jerry and Red Chalk (Main Basin) Lakes. The model was recalibrated for anoxic lakes (on Red Chalk Lake, East Basin) resulting in an adjustment of the settling velocity of TP from 12.4 m yr^{-1} to 7.2 m yr^{-1} .

An existing model for prediction of concentration of TP in lakes at spring turnover was refined to allow prediction of TP in any season and to incorporate the relation between the concentration of TP in the lake and that in the lake outlet. In lakes with no anthropogenic input of TP, the predicted and observed mean (1976-1980) concentrations of TP during the ice-free season agreed very closely ($9.3 \mu\text{g L}^{-1}$ vs. $9.3 \mu\text{g L}^{-1}$ in Jerry Lake and $5.5 \mu\text{g L}^{-1}$ vs. $5.6 \mu\text{g L}^{-1}$ in the Main Basin of Red Chalk Lake), giving confidence in the use of the model in lakes with anthropogenic inputs of TP. The incorporation of thermal stratification in the model did not improve its predictive capability.

Using the revised TP retention and concentration models, the predicted and observed concentrations of TP were compared for the four "A" lakes with shoreline development and the fraction of the potential anthropogenic TP input entering the lakes was calculated. For Chub, Blue Chalk and Dickie Lakes, the predicted and observed TP levels agreed to within $1 \mu\text{g L}^{-1}$ if it was assumed that 100% of the anthropogenic TP supply entered the lakes. However, for Harp Lake, only 20% of the anthropogenic TP supply must enter the lake for the observed and predicted TP levels to agree. This difference is likely due to differences in the TP retention capability of the soils surrounding the lake but this was not quantified.

Models were developed on the "A" and "B" lake data to predict the phytoplankton biomass, 'corrected' phytoplankton biomass (excluding diatoms) and the concentration of chlorophyll *a* (a correlate of phytoplankton

biomass) during the ice-free season from epilimnetic concentrations of algal nutrients (1976-1980 mean values). Chlorophyll *a* and 'corrected' phytoplankton biomass could both be predicted from the concentration of TP and the ratio of the concentrations of TN to TP. The phytoplankton biomass was related to the concentration of total organic N. The inclusion of total zooplankton biomass or herbivorous zooplankton biomass in the relations did not improve their predictive ability.

Water clarity (as measured by Secchi depth) of the "A" and "B" lakes during the ice-free season was controlled by the dissolved organic carbon content (or colour) of the lakes. Chlorophyll *a* concentration and phytoplankton biomass were not strongly correlated with water clarity.

The models of Cornett and Rigler (1979) and Welch and Perkins (1979) for prediction of areal hypolimnetic oxygen rates were tested. The Cornett-Rigler model, when recalibrated with data collected in this study, provided the closest agreement between predicted and observed depletion rates.

Total crustacean zooplankton biomass and herbivorous zooplankton biomass of the "A" and "B" lakes (1977-1979 mean values) could be predicted from the whole-lake concentrations of TP during the summer stratified period. These predictions may be useful in future efforts to model fish biomass.

Finally, a step-by-step methodology for predicting the effects of shoreline development on the trophic status of lakes is provided.

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APPENDIX

Tables A1 – A6.

Annual water balances for six study lakes.

Tables A7 – A12.

Major ion mass balances for six study lakes.

Tables A13 – A18.

Total phosphorus mass balances for six study lakes.

Table A1. Annual water balances for Harp Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3\text{m}^3\text{yr}^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	605	24.7	606	22.2	763	20.7	630	17.2	2,604	20.8
Inlet 3	105	4.3	114	4.2	144	3.9	144	3.9	507	4.1
3A	58.8	2.4	132	4.8	141	3.8	119	3.3	451	3.6
4	458	18.7	571	20.9	911	24.7	771	21.1	2,711	21.7
5	864	35.3	851	31.2	1,130	30.6	1,420	38.8	4,265	34.1
6	43.2	1.8	56.5	2.1	82	2.2	69.7	1.9	251	2.0
6A	52.0	2.1	63.1	2.3	83	2.3	86.6	2.4	285	2.3
Ungauged	259	10.6	333	12.2	435	11.8	423	11.6	1,450	11.6
Total	2,445	99.9	2,727	99.9	3,689	100.0	3,663	100.2	12,524	100.0
Loss terms										
Outlet	1,770	78.4	2,530	87.1	3,550	88.7	3,400	88.9	11,250	86.6
Evaporation	488	21.6	374	12.9	452	11.3	424	11.1	1,738	13.4
Total	2,258	100.0	2,904	100.0	4,002	100.0	3,824	100.0	12,988	100.0
lake storage	0	–	98.3	–	16.1	–	–68.2	–	46.2	–
Balance¹	187	8.0	–275	–9.8	–329	–8.6	–92.8	–2.5	–510	–3.6

¹ Balance = supply – loss $\pm$ Δ storage

$$\% \text{ balance} = (\text{balance} \frac{\text{supply} + \text{loss}}{2}) 100$$

Table A2. Annual water balances for Jerry Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3\text{m}^3\text{yr}^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	453	12.1	454	11.5	571	10.6	471	9.2	1,949	10.7
Inlet 1	22.9	0.6	30.5	0.8	38.5	0.7	37	0.7	129	0.7
3	2,660	71.0	2,740	69.6	3,800	70.6	3,660	71.2	12,860	70.6
4	122	3.3	141	3.6	201	3.7	209	4.1	673	3.7
Ungauged	491	13.1	570	14.5	771	14.3	763	14.8	2,595	14.3
Total	3,749	100.1	3,936	100.0	5,382	99.9	5,140	100.0	18,206	100.0
Loss terms										
Outlet	3,590	90.3	3,840	93.1	5,520	94.1	4,460	93.2	17,410	92.8
Evaporation	386	9.7	286	6.9	348	5.9	328	6.8	1,348	7.2
Total	3,976	100.0	4,126	100.0	5,868	100.0	4,788	100.0	18,758	100.0
lake storage	46.6		29.1		-15.0		-71.1		-57.0	
Balance ¹	-274	-7.1	-219	-5.4	-471	-8.4	+423	8.5	-495	-2.7

¹ Balance = supply - loss $\pm$ Δ storage.

$$\% \text{ balance} = (\text{balance} / \frac{\text{supply} + \text{loss}}{2}) 100$$

Table A3. Annual water balances for Chub Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3\text{m}^3\text{yr}^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	265	22.2	263	17.9	435	25.0	380	21.0	1,343	21.6
Inlet 1	206	17.3	216	14.7	287	16.5	260	14.4	969	15.6
2	453	38.0	649	44.1	638	36.7	763	42.2	2,503	40.3
Ungauged	269	22.6	343	23.3	378	21.8	405	22.4	1,395	22.5
Total	1,193	100.1	1,471	100.0	1,738	100.0	1,808	100.0	6,210	100.0
Loss terms										
Outlet	863	76.9	1,360	88.5	1,640	88.4	1,800	89.3	5,663	86.7
Evaporation	260	23.1	176	11.5	216	11.6	215	10.7	867	13.3
Total	1,123	100.0	1,536	100.0	1,856	100.0	2,015	100.0	6,530	100.0
lake storage	-15.1		24.2		19.0		-30.0		-1.9	
Balance ¹	85.1	7.3	-89.2	-5.9	-137	-7.6	-177	-9.3	-318	-5.0

¹ Balance = supply - loss $\pm$ Δ storage

$$\% \text{ balance} = (\text{balance} / \frac{\text{supply} + \text{loss}}{2}) 100$$

Table A4. Annual water balances for Dickie Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3\text{m}^3\text{yr}^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	791	32.1	751	28.6	1,258	35.6	1,100	29.5	3,900	31.6
Inlet 5	175	7.1	159	6.1	190	5.4	226	6.1	750	6.1
6	98.7	4.0	118	4.5	151	4.3	145	3.9	513	4.2
8	187	7.6	230	8.8	218	6.2	400	10.7	1,035	8.4
10	353	14.3	358	13.6	457	12.9	495	13.3	1,663	13.5
11	325	13.2	424	16.2	548	15.5	548	14.7	1,845	15.0
Ungauged	533	21.6	584	22.3	709	20.1	811	21.8	2,637	21.4
Total	2,463	99.9	2,624	100.1	3,531	100.0	3,725	100.0	12,340	100.2
Loss terms										
Outlet	1,900	72.6	2,200	78.9	2,690	79.8	2,910	82.8	9,700	78.9
Evaporation	716	27.4	580	21.1	681	20.2	605	17.2	2,582	21.1
Total	2,616	100.0	2,780	100.0	3,371	100.0	3,515	100.0	12,280	100.0
lake storage	0		105		18.6		-144		-20.5	
Balance ¹	-153	-6.0	-261	-9.7	141	4.1	354	9.8	81.5	0.7

¹ Balance = supply - loss $\pm$ Δ storage

$$\% \text{ balance} = (\text{balance} / \frac{\text{supply} + \text{loss}}{2}) 100$$

Table A5. Annual water balances for Blue Chalk Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3\text{m}^3\text{yr}^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	407	49.1	404	45.3	667	52.9	583	48.9	2,061	49.4
Inlet 1	85.6	10.3	53.2	6.0	62.6	5.0	108	9.1	309	7.4
Ungauged	336	40.5	435	48.7	530	42.1	500	42.0	1,801	43.2
Total	829	99.9	892	100.0	1,260	100.0	1,191	100.0	4,171	100.0
Loss terms										
Outlet	644	63.2	713	71.2	962	74.1	1,030	77.6	3,349	72.1
Evaporation	375	36.8	288	28.8	337	25.9	297	22.4	1,297	27.9
Total	1,019	100.0	1,001	100.0	1,299	100.0	1,327	100.0	4,646	100.0
lake storage	-113		77.1		21.2		-16.8		-31.1	
Balance ¹	-77	-8.3	-186	-19.7	-60.2	-4.7	-119	-9.5	-444	-10.1

¹ Balance = supply - loss $\pm$ Δ storage

$$\% \text{ balance} = (\text{balance} / \frac{\text{supply} + \text{loss}}{2}) 100$$

Table A6. Annual water balances for Red Chalk Lake for 1976-77, 1977-78, 1978-79, 1979-80 and 4 year balance for 1976-80. Individual supply and loss terms in $10^3\text{m}^3\text{yr}^{-1}$ and as % total supply or % total loss.

	1976-1977		1977-1978		1978-1979		1979-1980		1976-1980	
	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	$10^3\text{m}^3\text{yr}^{-1}$	%	10^3m^3	%
Supply terms										
Precipitation	468	17.5	465	14.9	768	18.9	671	17.0	2,372	17.2
Inlet 1	648	24.3	695	22.3	872	21.5	885	22.4	3,100	22.4
2	94	3.5	120	3.9	175	4.3	149	3.8	538	3.9
3	368	13.9	507	16.3	550	13.6	533	13.5	1,958	14.2
4	163	6.1	250	8.0	281	6.9	265	6.7	959	7.0
Blue Chalk outlet	644	24.1	713	22.9	962	23.7	1,030	26.0	3,349	24.3
Ungauged	286	10.7	369	11.8	451	11.1	425	10.7	1,531	11.1
Total	2,671	100.1	3,119	100.1	4,059	100.0	3,958	100.1	13,807	100.1
Loss terms										
Outlet	2,490	85.8	2,940	90.1	4,040	91.6	3,970	91.3	13,440	90.0
Evaporation RC main	320	11.0	249	7.6	283	6.4	291	6.7	1,143	7.7
Evaporation RC east	93.3	3.2	75.1	2.3	88.8	2.0	85.4	2.0	343	2.3
Total	2,903	100.0	3,264	100.0	4,412	100.0	4,346	100.0	14,925	100.0
lake storage	0		64.9		18.2		-12.0		71.1	
Balance ¹	-232	-8.3	-210	-6.6	-371	-8.8	-376	-9.1	-1,189	-8.3

¹ Balance = supply - loss $\pm \Delta$ storage

$$\% \text{ balance} = (\text{balance} / \frac{\text{supply} + \text{loss}}{2}) 100$$

Table A7. Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- mass balances for Harp Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	6,960	8,500	10,600	10,300	36,360
Δ Storage	0	+316	+55	-228	+143
Output	5,720	9,100	12,500	11,000	38,320
Δ	+1,240	-916	-1,955	-472	-2,103
%	+21.7	-10.1	-15.6	-4.3	-5.5
Mg^{2+}					
Input	2,140	2,410	2,820	2,620	9,990
Δ Storage	0	+97	+15	-60	+52
Output	1,790	2,610	3,400	2,880	10,680
Δ	+350	-297	-595	-200	-742
%	+16.4	-11.4	-17.5	-6.9	-7.0
Na^+					
Input	2,140	2,690	3,550	2,910	11,290
Δ Storage	0	+97	+17	-70	+44
Output	1,920	2,750	3,860	3,650	12,180
Δ	+220	-157	-327	-670	-934
%	+11.5	-5.7	-8.5	-18.4	-7.7
K^+					
Input	1,170	1,130	1,160	1,500	4,960
Δ Storage	0	+55	+9	-38	+26
Output	1,050	1,510	1,900	1,990	6,450
Δ	+120	-435	-749	-452	-1,516
%	+11.4	-28.8	-39.4	-22.7	-23.5
Cl^-					
Input	1,780	2,140	2,530	2,620	9,070
Δ Storage	0	+71	+12	-55	+28
Output	1,350	2,050	3,020	3,050	9,470
Δ	+430	+19	-502	-375	-428
%	+24.2	+0.9	-16.6	-12.3	-4.5

Table A8. Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- mass balances for Jerry Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	12,500	15,200	19,100	17,500	64,300
Δ Storage	+168	+103	-57	-252	-38
Output	12,200	14,700	21,500	15,600	64,000
Δ	+132	+397	-2,343	+2,152	+338
%	+1.1	+2.7	-10.9	+13.8	+0.5
Mg^{2+}					
Input	4,330	4,740	5,480	4,560	19,110
Δ Storage	+56	+34	-18	-71	+1
Output	4,080	4,720	6,100	4,220	19,120
Δ	+194	-14	-602	+411	-11
%	+4.8	-0.3	-9.9	+9.7	-0.1
Na^+					
Input	3,400	4,130	5,300	4,450	17,280
Δ Storage	+40	+27	-16	-64	-13
Output	3,220	4,150	7,040	4,280	18,690
Δ	+140	-47	-1,724	+234	-1,397
%	+4.4	-1.1	-24.5	+5.5	-7.5
K^+					
Input	1,880	2,110	2,420	2,610	9,020
Δ Storage	+27	+16	-8	-37	-2
Output	2,010	2,170	2,650	2,400	9,230
Δ	-157	-76	-222	+247	-208
%	-7.8	-3.5	-8.4	+10.3	-2.3
Cl^-					
Input	2,600	2,570	3,400	2,780	11,350
Δ Storage	+24	+16	-8	-41	-9
Output	1,990	2,180	3,060	2,360	9,590
Δ	+586	+374	+348	+461	+1,769
%	+29.5	+17.2	+11.4	+19.5	+18.5

Table A9. Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- mass balances for Dickie Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	5,460	5,840	7,070	7,560	25,930
Δ Storage	0	+289	+52	-395	-54
Output	4,980	6,140	7,200	8,040	26,360
Δ	+480	-589	-182	-85	-376
%	+9.6	-9.6	-2.5	-1.1	-1.4
Mg^{2+}					
Input	1,300	1,340	1,550	1,630	5,820
Δ Storage	0	+72	+12	-84	0
Output	1,220	1,490	1,660	1,590	5,960
Δ	80	-222	-122	+124	-140
%	+6.6	-14.9	-7.4	+7.8	-2.3
Na^+					
Input	1,630	2,190	2,310	2,100	8,230
Δ Storage	0	+89	+17	-123	-17
Output	1,560	1,940	2,470	2,530	8,500
Δ	+70	+161	-177	-307	-253
%	+4.5	+8.3	-7.2	-12.1	-3.0
K^+					
Input	999	903	943	1,130	3,975
Δ Storage	0	+48	+8	-60	-4
Output	831	995	1,080	1,110	4,016
Δ	+168	-140	-145	+80	-37
%	+20.2	-14.1	-13.4	+7.2	-0.9
Cl^-					
Input	2,380	2,690	2,850	3,040	10,960
Δ Storage	0	+92	+16	-125	-17
Output	1,690	2,120	2,420	2,600	8,830
Δ	+690	+478	+414	+565	+2,147
%	+40.8	+22.6	+17.1	+21.7	+24.3

Table A10. Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- mass balances for Chub Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	3,610	4,250	4,470	4,600	16,930
Δ Storage	-43	+72	+58	-87	0
Output	2,580	4,190	4,760	4,870	16,400
Δ	+1,073	-12	-348	-183	+530
%	+41.6	-0.3	-7.3	-3.8	+3.2
Mg^{2+}					
Input	927	1,140	1,030	1,040	4,137
Δ Storage	-11	+18	+14	-19	+2
Output	657	1,060	1,120	1,100	3,937
Δ	+281	+62	-104	-41	+198
%	+42.8	+5.9	-9.3	-3.7	+5.0
Na^+					
Input	904	1,330	1,380	1,250	4,864
Δ Storage	-11	+19	+15	-22	+1
Output	634	1,110	1,580	1,290	4,614
Δ	+281	+201	-215	-18	+249
%	+44.3	+18.1	-13.6	-1.4	+5.4
K^+					
Input	721	635	608	691	2,655
Δ Storage	-7	+11	+8	-12	0
Output	392	632	646	734	2,404
Δ	+336	-8	-46	-31	+251
%	+85.7	-1.3	-7.1	-4.2	+10.4
Cl^-					
Input	1,170	1,240	1,150	1,230	4,790
Δ Storage	-9	+15	+11	-18	-1
Output	538	955	987	1,030	2,510
Δ	+641	+270	+152	+218	+1,281
%	+119.1	+28.3	+15.4	+21.2	+36.5

Table A11. Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- mass balances for Blue Chalk Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	1,680	1,840	2,010	1,970	7,500
Δ Storage	-319	+224	+64	-48	-79
Output	1,920	2,230	3,120	2,870	10,140
Δ	+79	-614	-1,174	-852	-2,561
%	+4.1	-27.5	-37.6	-29.7	-25.3
Mg^{2+}					
Input	386	434	433	411	1,664
Δ Storage	-91	+61	+16	-11	-25
Output	513	567	743	677	2,500
Δ	-36	-194	-326	-255	-811
%	-7.0	-34.2	-43.9	-37.7	-32.4
Na^+					
Input	481	636	648	535	2,300
Δ Storage	-95	+64	+18	-14	-27
Output	547	616	967	890	3,020
Δ	+29	-44	-337	-341	-693
%	+5.3	-7.1	-34.9	-38.3	-23.0
K^+					
Input	292	271	227	287	1,077
Δ Storage	-43	+30	+8	-6	-11
Output	241	300	439	422	1,402
Δ	+94	-59	-220	-129	-314
%	+39.0	-19.7	-50.1	-30.6	-22.4
Cl^-					
Input	421	436	439	416	1,712
Δ Storage	-48	+36	+8	-8	-12
Output	426	331	474	477	1,708
Δ	+43	+69	-43	-53	+16
%	+10.1	+20.9	-9.1	-11.1	+0.9

Table A12. Ca^{2+} , Mg^{2+} , Na^+ , K^+ and Cl^- mass balances for Red Chalk Lake for 4 12-month periods and the total 4-year period. Input includes that from all gauged inflows, from the ungauged area and from atmospheric deposition, output consists of loss via outflow, and Δ storage represents the change in mass in the lake as a result of change in lake water level. The difference (Δ) is expressed as the surplus or deficit relative to the output, and the % is calculated relative to the output.

	1976-77	1977-78	1978-79	1979-80	1976-80
Ca^{2+}					
Input	7,120	9,030	10,500	9,280	35,930
Δ Storage	0	+190	+56	-35	+211
Output	6,810	9,100	12,400	10,500	38,810
Δ	+310	-260	-1,956	-1,185	-3,091
%	+4.6	-2.9	-15.8	-11.3	-8.0
Mg^{2+}					
Input	1,860	2,340	2,490	2,130	8,820
Δ Storage	0	+55	+15	-8	+62
Output	1,950	2,720	3,280	2,690	10,640
Δ	-90	-435	-805	-552	-1,882
%	-4.6	-16.0	-24.5	-20.5	-17.7
Na^+					
Input	1,940	2,620	3,120	2,650	10,330
Δ Storage	0	+52	+15	-10	+57
Output	1,780	2,350	3,060	3,210	10,400
Δ	+160	+218	+45	-550	-127
%	+9.0	+9.3	+1.5	-17.1	-1.2
K^+					
Input	1,060	1,280	1,240	1,320	4,900
Δ Storage	0	+29	+8	-5	+32
Output	1,030	1,390	1,640	1,680	5,740
Δ	+30	-139	-408	-355	-872
%	+2.9	-10.0	-24.9	-21.1	-15.2
Cl^-					
Input	1,620	1,730	1,950	1,810	7,110
Δ Storage	0	+30	+9	-6	+33
Output	1,190	1,390	1,870	1,730	6,180
Δ	+430	+310	+71	+86	+897
%	+36.1	+22.3	+3.8	+5.0	+14.5

Table A13. Summary of total phosphorus mass balance for Harp Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 6 gauged subwatersheds. Results are in kg yr⁻¹, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 3	3.29 (4.1)	2.47 (4.0)	3.26 (4.8)	2.84 (4.8)	11.9 (4.4)
Inflow 3A	0.47 (0.6)	0.88 (1.4)	0.73 (1.1)	0.74 (1.2)	2.82 (1.0)
Inflow 4	7.44 (9.2)	10.4 (16.8)	11.9 (17.5)	10.9 (18.4)	40.6 (15.0)
Inflow 5	23.9 (29.5)	17.0 (27.5)	23.9 (35.1)	21.2 (35.7)	86.0 (31.8)
Inflow 6	0.75 (1.0)	0.81 (1.3)	0.99 (1.5)	0.88 (1.5)	3.43 (1.3)
Inflow 6A	0.68 (0.8)	0.73 (1.2)	0.84 (1.2)	1.84 (3.1)	4.09 (1.5)
Ungauged	5.39 (6.7)	5.07 (8.2)	6.21 (9.1)	6.34 (10.7)	23.0 (8.5)
Total terrestrial	41.9 (51.7)	37.4 (60.5)	47.8 (70.3)	44.7 (75.3)	171.8 (63.6)
Atmosphere	39.1 (48.3)	24.4 (39.5)	20.2 (29.7)	14.6 (24.6)	98.3 (36.4)
Total measured	81.0 (100.0)	61.8 (100.0)	68.0 (100.0)	59.4 (99.9)	270.2 (100.0)
Development ¹	79.8	81.2	82.5	82.5	326.0
Outflow	14.3	16.9	27.2	26.4	84.8

¹ Estimated in Table 32.

Table A14. Summary of total phosphorus mass balance for Dickie Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 5 gauged subwatersheds. Results are in kg yr⁻¹, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 5	6.45 (6.0)	4.85 (5.6)	3.85 (5.1)	6.09 (7.0)	21.2 (5.9)
Inflow 6	3.78 (3.5)	5.89 (6.8)	7.75 (10.2)	11.5 (13.1)	28.9 (8.1)
Inflow 8	3.33 (3.1)	3.93 (4.5)	2.32 (3.1)	4.97 (5.7)	14.6 (4.1)
Inflow 10	15.3 (14.1)	11.1 (12.8)	8.79 (11.6)	10.1 (11.5)	45.3 (12.6)
Inflow 11	7.05 (6.5)	8.87 (10.2)	7.29 (9.6)	9.29 (10.6)	32.5 (9.1)
Ungauged	18.1 (16.7)	18.2 (21.0)	17.5 (23.1)	25.3 (28.9)	79.1 (22.1)
Total terrestrial	54.0 (49.8)	52.8 (60.8)	47.5 (62.8)	67.3 (76.9)	221.6 (61.8)
Atmosphere	54.4 (50.2)	34.0 (39.2)	28.2 (37.3)	20.3 (23.2)	136.9 (38.2)
Total measured	108.4 (100.0)	86.8 (100.0)	75.7 (100.1)	87.5 (100.1)	358.4 (100.0)
Development ¹	75.3	77.2	79.1	79.1	310.7
Outflow	22.8	27.3	33.2	31.7	115.0

¹ Estimated in Table 32.

Table A15. Summary of total phosphorus mass balance for Chub Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 2 gauged subwatersheds. Results are in kg yr⁻¹, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	1.55 (3.8)	2.24 (6.2)	2.30 (7.1)	2.11 (5.4)	8.20 (5.5)
Inflow 2	14.9 (36.2)	15.7 (43.3)	14.2 (43.8)	21.2 (54.6)	66.0 (44.4)
Ungauged	6.02 (14.6)	6.67 (18.4)	6.20 (19.1)	8.51 (21.9)	27.4 (18.4)
Total terrestrial	22.5 (54.6)	24.6 (67.8)	22.7 (70.0)	31.8 (81.9)	101.6 (68.3)
Atmosphere	18.8 (45.6)	11.7 (32.2)	9.74 (30.0)	7.02 (18.1)	47.3 (31.8)
Total measured	41.2 (100.2)	36.3 (100.0)	32.5 (100.0)	38.8 (100.0)	148.8 (100.1)
Development ¹	2.57	2.70	2.82	2.82	10.9
Outflow	8.43	13.9	15.0	17.9	55.2

¹ Estimated in Table 32.

Table A16. Summary of total phosphorus mass balance for Blue Chalk Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 5 gauged subwatersheds of Blue Chalk and Red Chalk Lakes. Results are in kg yr⁻¹, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	0.43 (1.3)	0.47 (1.9)	0.29 (1.4)	0.99 (5.5)	2.18 (2.2)
Ungauged	4.56 (13.5)	6.30 (25.4)	5.70 (27.3)	6.26 (34.7)	22.8 (23.4)
Total terrestrial	4.99 (14.8)	6.77 (27.3)	5.99 (28.7)	7.25 (40.2)	25.0 (25.6)
Atmosphere	28.8 (85.2)	18.0 (72.6)	14.9 (71.3)	10.8 (59.8)	72.5 (74.4)
Total measured	33.8 (100.0)	24.8 (99.9)	20.9 (100.0)	18.0 (100.0)	97.5 (100.0)
Development ¹	5.66	5.66	5.66	5.66	22.6
Outflow	3.49	4.86	5.57	6.51	20.4

¹ Estimated in Table 32.

Table A17. Summary of total phosphorus mass balance for Red Chalk Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 5 gauged subwatersheds of Blue Chalk and Red Chalk Lakes. Results are in kg yr⁻¹, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	6.23 (10.8)	9.32 (16.6)	9.58 (18.9)	12.5 (25.2)	37.6 (17.5)
Inflow 2	1.31 (2.3)	1.69 (3.0)	2.01 (4.0)	1.67 (3.4)	6.68 (3.1)
Inflow 3	4.31 (7.5)	7.83 (13.9)	6.44 (12.7)	6.20 (12.5)	24.8 (11.6)
Inflow 4	5.32 (9.2)	6.41 (11.4)	5.19 (10.2)	4.93 (9.9)	21.9 (10.2)
Blue Chalk outflow	3.49 (6.0)	4.86 (8.6)	5.57 (11.0)	6.51 (13.1)	20.4 (9.5)
Ungauged	3.88 (6.7)	5.35 (9.5)	4.84 (9.5)	5.32 (10.7)	19.4 (9.0)
Total terrestrial	24.5 (42.5)	35.5 (63.1)	33.6 (66.2)	37.1 (74.9)	130.7 (61.0)
Atmosphere	33.2 (57.4)	20.7 (36.8)	17.2 (33.9)	12.4 (25.0)	83.5 (38.9)
Total measured	57.8 (99.9)	56.2 (99.9)	50.8 (100.1)	49.6 (99.9)	214.4 (99.9)
Development ¹	0.60	0.60	0.60	0.60	2.40
Outflow	12.2	20.8	24.6	26.3	83.9

¹ Estimated in Table 32.

Table A18. Summary of total phosphorus mass balance for Jerry Lake. Input from the ungauged area was calculated on the basis of the measured total phosphorus gross export (yield per unit area) of the 3 gauged subwatersheds. Results are in kg yr⁻¹, figures in brackets are per cent of total measured input contributed by source.

Source	1976-77	1977-78	1978-79	1979-80	1976-80
Inflow 1	0.29 (0.3)	0.29 (0.4)	0.38 (0.4)	0.38 (0.5)	1.34 (0.4)
Inflow 3	71.4 (63.6)	52.5 (66.0)	74.6 (72.9)	59.3 (72.6)	257.8 (68.6)
Inflow 4	1.92 (1.7)	1.23 (1.6)	1.93 (1.9)	1.92 (2.4)	7.00 (1.9)
Ungauged	9.42 (8.4)	7.19 (9.0)	10.2 (10.0)	9.13 (11.2)	35.9 (9.6)
Total terrestrial	83.0 (73.9)	61.2 (77.0)	87.1 (85.1)	70.7 (86.5)	302.0 (80.4)
Atmosphere	29.3 (26.1)	18.3 (23.0)	15.2 (14.9)	10.9 (13.3)	73.7 (19.6)
Total measured	112.3 (100.0)	79.5 (100.0)	102.3 (100.0)	81.7 (99.8)	375.8 (100.0)
Development ¹	0.0	0.0	0.0	0.0	0.0
Outflow	29.9	27.7	45.9	47.4	150.9

¹ Estimated in Table 32.